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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. 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 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 the existing database of economic values. The focus of the paper is on the European policy arena, but most of the issues discussed apply equally to other locations.

We approach the question in three ways. First, by reviewing existing European legislative drivers for increased use of valuation in coastal and marine policy; second, by reviewing the existing body of evidence on ecosystem and biodiversity values related to the coastal and marine environment in the EU and third, by considering 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.

2. Legislative drivers.

Amongst major new pieces of marine legislation in Europe which require inputs of environmental valuation are the following. It is interesting to note that as European environmental policy has developed over the last 20 years, the need for the valuation of impacts of such legislation has become more explicit in the policy documents themselves. In earlier versions of the 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. 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.

Revised Bathing Waters Directive:

The revised Bathing Water Directive (2006/7/EC) will come into force in 2015, replacing and updating the current Directive (76/10/EEC). This 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 parameters 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.

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 (e.g. Marine Scotland Act, 2010). MPAs are now 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 ecosystem services (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.

Elsewhere, some member states have taken their own initiative to establish a broader definition of MPAs. In the UK for example, the Marine and Coastal Access Act 2009 committed the UK government to the delivery of an "ecologically coherent" network of MPAs to be established. Since then, a network of proposed sites in both England and Scotland has been announced. The UK Act has resulted in a substantial amount of economic analysis associated with its 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 terms for a network of sites, with by far the biggest single benefit being for enhanced greenhouse gas mitigation.

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, and this compared the likely benefit 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".

However, 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. On the 24 July 2014, 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 to show the relative benefits of future MPA designations.

Marine Strategy Framework Directive (MSFD)

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

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 ongoing assessments 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.

Other potential applications.

However, there are also many other instances where an improvement in the quality of policy design and implementation would likely result from the use of economic evidence. One example is the regulation of new deep-sea mining sites where natural recovery is likely to be slow, and where restoration costs are expected to be several orders of magnitude higher than for terrestrial mining activities due to difficult access and the remoteness of deep-sea sites (Van Dover et al., 2014). Another example is the spatial targeting of nutrient reduction programmes in estuaries, bays and sea lochs (fjords) where nutrient inputs are damaging biodiversity and some ecosystem services, but where there is significant spatial variation in the benefits and costs of nutrient reduction programmes (Ahtiainen et al, 2014). A third example would be the consenting and planning process for off-shore renewable energy developments (e.g. barrages, wind farms, wave and tidal power installations) where the inclusion of the monetary value of both positive and negative environmental impacts would improve the comprehensiveness of social cost-benefit analysis of such proposals (Landenberg and Dubgaard, 2007; Kruger et al, 2011, Furness et al 2013).

3. How comprehensive is the economic evidence base?

In this section, the quality and extent of existing studies of coastal and marine ecosystem values in Europe is reviewed (although we do not try to be comprehensive here). A number of environmental valuation methods have been employed to quantify the benefits associated with the introduction of marine environmental policies such as those mentioned in the previous section. These methods include stated preference approaches (contingent valuation and choice experiments), travel cost models, hedonic pricing approaches and production function approaches (Hanley and Barbier, 2009). The focus in this section is on stated preference studies, since those have been the most numerous. Benefit estimates from such studies can then be compared with the costs of implementing such policies in order to judge the overall social efficiency of the legislation and the desirability and targeting of “derogations” from uniform targets that in particular feature in the EU Bathing Waters Directive, the EU Water Framework Directive and the EU Marine Strategy Framework Directive (Hynes et al. 2013, Hussain et al 2009; Hanley et al. 2006 and Georgiou et al. 1998). Valuation can also be used to assess the distributional implications of marine and coastal legislation, in terms of how benefits and costs vary across stakeholders.

Valuation studies concerning EU marine and coastal environmental policies include Georgiou et al. (2004) who undertook a contingent valuation (CV) examining the benefits of coastal water bodies meeting the EC Directive on Bathing Water (CEC, 1976). Elsewhere, Östberg et al (2012) examined the feasibility of using the CV method for estimating willingness to pay (WTP) for marine environmental improvements. Using a case study in two Swedish coastal areas, the authors examined whether respondents were able to understand and attach a monetary value to marine policy–determined scenarios. The tested scenarios were based on improving coastal water quality according to the EU Water Framework Directive and reducing noise and littering in a Swedish archipelago setting.

The Choice Experiment (CE) method is a popular valuation methodology that has been applied in a number of recent studies assessing changes to the quality of marine and coastal environmental resources that have been brought about under new polices (Eggert and Olsson, 2009; Kosensius, 2010 and McVittie and Moran , 2010). Eggert and Olsson (2009) used a CE to represent marine water quality by three different attributes; coastal cod stock level, bathing water quality, and biodiversity level. The CE was carried out amongst residents on the Swedish west coast to estimate the economic benefits of improved coastal water quality. The authors found that the highest values were placed by respondents on preventing further depletion of marine biodiversity and to improve Swedish cod stocks. The ecosystem service values resulting from changes to the Bathing Waters Directive were examined using a choice experiment by 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.

Also using the CE approach, Kosensius (2010) examined the preference heterogeneity of the general public regarding water quality attributes and provides welfare estimates for three nutrient-reduction scenarios that would improve the water quality in the Gulf of Finland. In the choice experiment, improvements were described in terms of four attributes: water clarity, abundance of coarse fish (fresh water fish other than the game fish: trout salmon and char), status of bladder wrack (*Fucus vesiculosus*, a type of brown seaweed), and mass occurrences of blue-green algal (cyanobacterial) blooms. The models estimated from choice responses were then used to examine the welfare impact associated with three nutrient reduction scenarios of different intensities.

A Choice Experiment study related to the UK Marine and Coastal Access Act 2009 was designed to inform the draft legislation, focussed on two types of benefit from designation as an MPA (conservation of biodiversity; ecosystem services provided by the designated sites - CO_2 sequestration, water treatment and recreation) along with 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) benefits of designation were greater than the PV costs, and about 10 times higher.

A recent UK National Ecosystem Assessment case study estimated the economic values of cultural ecosystem services to recreational users of MPAs (Jobstvogt et al., 2014; Kenter et al., 2013; Kenter et al., 2014). A combination of attribute-based CV and CE based on travel-cost was proposed to assess non-use and use values within a single survey (Jobstvogt et al., 2014; Kenter et al. 2013). Follow-on workshops highlighted the diminishing effect on WTP that group deliberation had in the elicitation process (Kenter 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.

While the initial assessments conducted by each EU member state as part of the legal requirements under the MSFD were completed last year, the specific requirement for EU Member States to carry out “an economic analysis of the cost of degradation of the marine environment” as an integral part of their initial assessments was ignored by all but one member state. The initial assessment carried out by Ireland (Department of the Environment, Community and Local Government, 2013) included a Choice Experiment that was employed to estimate the value that Irish residents have for the non-market ecosystem service 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 (and combined) to generate the attributes used in this CE. The welfare impact of a change in the marine environmental attributes associated with 3 possible future marine environmental degradation scenarios was 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 large (Norton and Hynes, 2014).

As one moves further offshore, the literature related to the valuation of marine ecosystem services becomes scarcer. This may be due to the fact that 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). In addition, changes that impact deep ocean circulation may take many tens of years to become relevant (Tyrrell 2011) and may affect surface ES flows such as fisheries and biogeochemical transformations. It follows that our ability to link changes in these functions as a result of anthropogenic pressures to effects on human well-being, are limited. Armstrong et al. (2012) present a categorisation and synthesis of deep-sea ecosystem goods and services, and review the current state of human knowledge about these services, the possible methods of their valuation, and possible steps forward in its implementation. Jobstvogt et al (2014a) discuss the problems of estimating economic values for some of these deep-sea ecosystem service flows. As pointed out by Jobstvogt et al. (2014b), one of the biggest challenges of attaching economic values to deep-sea ES and biodiversity relates to the unfamiliarity of the general public with the deep-sea environment. Deeper waters are prone to under-valuation and lack of appreciation as a result of their remoteness and a perceived lack of relevance to everyday life. This makes the use of stated preference methods such as CE and CV to estimate values for deep-sea biodiversity much more difficult.

The above-mentioned difficulties aside, a number of studies have attempted to value deep-sea ecosystem service benefit values. Jobstvogt et al. (2014b) carried out a CE survey that asked Scottish households for their WTP for additional MPAs in the Scottish deep sea. The experiment focused on the elicitation of economic values for two attributes of deep sea; (i) the existence value for deep-sea species and (ii) the option-use value of deep-sea organisms as a source for future medicinal products. Elsewhere, Ressurreição et al. (2011) used a CV method to estimate the willingness to pay of residents and visitors to two Azorean islands to avoid losses in the number of marine species in the waters around the Azores archipelago. The authors estimated the marginal value associated with increased levels of species loss for mammals, fish, algae, birds and invertebrates. Earlier work by Glenn et al. (2010) estimated non-use values that the general public had for the protection of cold water corals in Irish waters. The results from this survey indicated a willingness to pay for cold water corals protection between €0-10 per person. Finally, Aanesen et al (2014) used choice modelling to study the values which Norwegians placed on alternative protection programmes for cold-water corals off the coast of Norway. They found that the habitat provision function of cold water corals for other marine life was particularly highly valued, whilst average WTP for a specific protection programme was around €200 per household per year.

The increasing use of ecosystem service benefit values in policy decisions has meant an increase in demand for 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 ecosystem service 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 ecosystems types. In the case of coastal and marine resources, examples of ecosystem service 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].

4. Is the economic value framework fit for purpose?

Economic values for ecosystem services need to be founded in the principles of applied welfare economics (Boadway and Bruce, 1984). This means that an ecosystem service 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. they enjoy watching sea birds). An indirect effect occurs when an ecosystem service "S" is used in the production of a benefit "X" which itself appears in the utility function. Thus, estuaries contribute recreational fishing opportunities which allow the "production" of recreational fishing trips along with inputs of leisure time, boats, gear etc. People then derive utility from fishing trips. An indirect benefit also occurs when the ecosystem service contributes to a service 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 (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 ecosystem service supply and a change in human well-being clearly depends on which kind

[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

of ecosystem service 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 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 ecosystem services set out in Bateman et al (2011) and UKNEA (2011) insists that only final impacts on human well-being be counted as economic benefits, to avoid double accounting; and that the contribution of ecosystem services to benefits be separated out 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" ecosystem services (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.

Is this a novel perspective on environmental values? Probably not. The distinction between the direct and indirect economic values of the environment is not at all new (Freeman, 1979), whilst the idea of valuing the environment as an input is also not new (Barbier, 2007). One way of thinking about the economic value of ecosystem services is that such service flows typically need to be combined with other inputs to produce benefits for individuals (Boyd and Banzaff, 2007). This means that production-function methods are often particularly suited to estimating the economic costs or benefits of a change in ecosystem services (ES) supply, since direct impacts on utility (which can be measured using stated or revealed preferences) only characterise a sub-set of the ways in which people interact with nature.

To be useable, the economic framework thus requires that (i) the direct and indirect links between utility and the condition 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 condition 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 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 in section 4. Condition (iii) may not be simple to meet, as the UKNEA (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 (for example, ocean acidification), 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). This makes it difficult to predict the effects on functioning and on consequent changes in ES flows and their value (Figure 1).

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). Whilst there have clearly been considerable gains in the sophistication with which these methods are applied and tested, and whilst the methods themselves have been extended (e.g. the growing use of choice experiments 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).

5. Is the science fit for purpose?

The link between environmental science and economic valuation is complex and requires clear understanding on both sides. The major scientific issues concerns the current “biodiversity-ecosystem function” (BEF) 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). 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 towards 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 system 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 (e.g. the NERC Biodiversity and Ecosystem Service Sustainability programme, 2012-2016). While the goals of these programmes would appear to serve the natural capital and ecosystem valuation agendas very well, the formulations are not yet ready and there is a possibility that a 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, the responses are 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 quantify 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 ecosystem services, considerably limits the extension of economic valuation studies.

6. Three examples.

In this section, we work our way through 3 examples of marine and coastal management issues. The intention is to illustrate the potential and limitations of economic valuation, and the extent to which the current scientific evidence base allows valuation to be undertaken. The case studies are not comprehensive analyses, but are intended to highlight different problems and potentials. The case studies are:

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

A. Deep-sea conservation.

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 through 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 social and 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 deep-sea ecosystem were identified and linked to ecosystem structures, processes and functions by Jobstvogt et al (2014a), and are shown here (Table 1). In this example, a structured elicitation of experts' ecological understanding helped to simplify and generalise these linkages for a potential uptake within ecosystem-based marine management.

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 anthropogenic impacts, and what this means for the supply of ES such as those listed in Table 1 (Armstrong et al, 2012). It is also worth noting the relative 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 consequence for future generations. This "deferment of consequence" may be a serious issue in the current management and valuation of the consequences of change.

Another problem relates to the ecosystem's remoteness and the non-expert's lack of experience with and awareness of the deep sea which makes the use of stated preference methods for valuing the benefits of protecting deep-sea areas difficult (Jobstvogt et al, 2014b). 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. The large predominance of regulating and intermediate services (climate regulation, habitat etc.) speaks for the high value that the ocean should possess: however, quantifying links to the supply of final ES is difficult.

From an economic valuation viewpoint, the lack of human interaction with the deep sea is problematic. Unlike with coastal systems, the vast majority of people cannot dive down to explore the deep sea. Lack of knowledge about the nature of the deep sea also complicates the use of stated preference methods. Such methods are, however, in principle able to estimate the non-use value for deep-sea areas. In the following we highlight some critical points to bear in mind when conducting such studies on ecosystem components that participants are likely to be unfamiliar with: (i) Respondents' low familiarity might result in constructed preferences and low reliability of WTP bids; (ii) balancing low familiarity with enough information might be beyond the scope of many survey tools; (iii) respondents might be indifferent to scope and scale of the attributes and the scenarios; (iv) lack of familiarity might translate directly into perceived irrelevance to the respondent; (v) benefits are poorly understood and their description without being persuasive highly challenging; (vi) it is difficult to think of survey attributes that are not overlapping, which can easily lead to poor survey design; (vii) perceptions on the status quo might differ among respondents and describing policy scenarios with uncertain outcome is problematic; (viii) there is a risk of confounding existence value with warm glow and ethical values; (ix) it is not straight forward to comprise the complexity of ecosystems within a survey and respondents might easily feel overwhelmed by the amount of information provided.

However, as noted in section 3, there are some examples of environmental valuation methods being applied to deep-sea protection scenarios. 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 (2014), again as introduced above. They tried to 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). No such significant treatment effect was found for those with below average scores.

B. **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. The management, protection and restoration of the natural habitats 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 through both wind and waves (Das et al 2013, Barbier et al., 2008), whilst 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. It should be noted that these combined and stand-alone soft engineering schemes require careful analysis of local conditions to ensure they are likely to be successful in reducing flood risks. Avoided cost methods, based on cost savings from lower spending on hard defences and land values analysed using a Ricardian model (for example) 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 ecosystem service flows that are less easy to value but should be include in a holistic assessment. Commonly cited services, in addition to coastal protection, include habitat provisioning (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 (Figure 2) while gaseous CO_2 variation can be determined at a larger scale using eddy covariance methodologies (Guo et al. 2009). Above- 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). It is also plausible that "carbon values" might be shared among salt marshes of similar characteristics, whereas aspects such as fisheries will be locally dependant.

In economic terms, coastal protection and associated land values may greatly outweigh other less direct services. However, this has not been fully explored, and the role of salt marshes in carbon sequestration is being more widely recognised (Luisetti et al, 2014). At present, valuing other ES than carbon sequestration and flood risk alleviation in saltmarshes may require similar approaches as in the deep sea case study (that is, by using CE and CV), 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.

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 in the linking of function to services and the required approaches to place an economic value on changes in these, a problem which is common to much Ecosystem Service valuation (Brander et al, 2013).

C. Location of new off-shore renewables

In October of 2014, the European Council adopted a target of reducing carbon emissions to 40% of 1990 levels by 2030. The major justification is the predicted impact of CO_2 on climate change, and this has spurred national and regional governments to seek alternative methods of energy generation. In terms of regional areas of Europe, Scotland is performing relatively well in increasing green energy production and has an ambitious target of supplying 100% of the demand for electricity from renewable sources by 2020. The natural climatic and oceanic conditions around the north of the UK provide great potential for wind, wave and current based off-shore energy production. It is estimated that the Scottish coast possesses 25% of exploitable European tidal power and 10% of Europe's wave power resources. However, the cost-benefit analysis of many of these potential developments is still incomplete and the science that is required to fully assess the impacts of increased investments in renewables often lags the political will to promote developments (Paterson et al 2012).

As indicated previously, the lack of a firm scientific basis makes the assessment of economic gains and loss is problematic. The example of offshore development in Scotland is useful. In 2004 it became a legal requirement[3] that all spatial plans would be subject to the Strategic Environmental Assessment Directive. The inherent logistic problems in assessing the status of an ecosystem often dominated by mobile species and advective transport of materials and which is often already heavily exploited for other ES provision makes the relative assessment of relative loss and gain very difficult. This requirement sits within the further EU framework of ecosystem assessment, policy and legislative efforts to achieve and maintain “Good Ecological Status” (GES) as demanded by the EU MSFD.

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, and when combined with the lack of a precise and coherent framework of requirements, methodology and data availability leaves economic assessment in a difficult position. Lack of knowledge means we are often “thinking on our feet”, developing new methods of assessment rapidly and often at the limits of current technology. Therefore, in terms of assessing ES flow and GES we are often required to employ subjective methodology such as expert opinion to providing a pragmatic solution to produce the best possible information and advice to support policy development. In terms of the offshore siting of energy generation systems a number of environmental factor have to be considered, a subset of which are given below:

- Marine Birds – including marine noise
- Marine Mammals – including marine noise
- Benthic Ecology
- Fish and Shellfish – including marine noise
- Commercial Fisheries
- Protected Sites and Species
- Seabed Contamination and Water Quality
- Electric 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 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) for the seabed adjacent to the Sullom Voe Oil Terminal in Shetland and spans over 30 years[3]. 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 requires a good knowledge of local and transient biodiversity and this is critical for the designation of the habits 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 offshore systems that

[3] *http://www.soteag.org.uk*

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 protected areas associated with developments such as wind farms to provide protection of habits form more damaging activities such as dredging must also be assessed. However, these problems are being assessed and the data gap is closing fast. Within the next decade, provided the impetus to deliver "clean" offshore energy continues, and funding is available, then the position will be greatly improved.

The emerging scientific evidence thus 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 very 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. 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 carry out a choice experiment with a sample of the UK public regarding the possible designation of a Marine Protected Area 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 4 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 invasives. 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.

7. Conclusions

Whilst developments in marine and coastal legislation 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. A recent paper by Guoa and Kildow (forthcoming) reported on a survey of key personnel from two US agencies - the National Oceanic and Atmospheric Administration (NOAA) and Environmental Protection Agency (EPA) - in which the authors attempted to find out if a gap exists between ecosystem service benefit valuation studies and their use in policy decisions related to the management of estuaries. Results of the survey showed that decision makers had limited knowledge about marine ecosystem valuation and its use, and varied opinions on the role and value of such studies in actual management. The authors note that restrictions that originate from administration agencies, such as tight budgets, limited access to economists and bias against social science research by natural scientists partially contributes to this lack of policy traction. Furthermore, they conclude that a major issue facing the use of such studies in the literature is that very few of such studies describe its practical usefulness for policy making or environmental management.

Whilst economic methods exist to allow the economic valuation of many ecosystem service and biodiversity benefits in coastal and marine areas, and whilst there is an increasing body of empirical evidence on these benefits, the discussion in the case studies makes it clear that problems remain. These are both conceptual and empirical. 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 biodiversity, ecosystem functions, and service flows. Uncertainties in the ecological evidence will necessarily have knock-on effects on the error margins of economic estimates. In principle, this is handle-able as long as these uncertainties are highlighted, both to respondents in stated preference exercises and to those using the valuation advice. However, especially when economic values are taken from valuation databases for benefit transfer, the value context as well as information on uncertainties are likely to be omitted.

Another limitation of economic valuation in regard to marine ecosystem service benefits 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 these 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. However, such studies have an important role in flagging up the potential economic values held by the average citizen, which are typically omitted from economic assessments due to the valuation challenges involved. And whilst there is certainly a benefit in integrating citizens' economic values into the decision making process, trade-offs have to be made on a scale between the accuracy of economic value estimates and their comprehensiveness.

Finally, we note that the classification system of ecosystem services which originated in the Millennium Ecosystem Assessment, namely classifying ES into provisioning, regulating, cultural and supporting services, does not fit very well with the economic valuation framework. This is for two reasons. First, as noted above, only those services which have impacts on human well-being should be subject to economic valuation. This means supporting ecosystem services should be excluded, and their implicit values only counted through their links with the delivery of provisioning, regulatory and cultural ES which are either directly linked (through being an argument in someone's utility function) or indirectly linked (by being part of an ecological production function which produces an output which is itself an argument in the utility function) to well-being. Second, this classification does not offer guidance to non-economists and/or policy makers and regulators as to which types of valuation approach are best suited to estimating the value of a particular service in a coastal or marine context.

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 choice modelling. Hattam et al (2015) provide a comprehensive account of how indicators reflecting changes in ecosystem function can be linked to ecosystem services 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." (page 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 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).

Questions also remain in relation to how human-induced ecosystem changes affect the provision of ecosystem services, how ecosystems interact to dictate the size of the impact on service provision, and how changes in the provision of such services ultimately affect the welfare of different groups in society. The integration of ecosystem service 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. Much closer collaboration at this level will be required in the future if an accurate picture is to be got of the causes and costs of potential marine environment degradation in any of Europe's regional.

While much has been done on the valuation side to produce more robust ecosystem service value estimates, insuring that these estimates are then routinely 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 ecosystem service research more effectively and to improve the understanding of the 'realities' of policymakers to economists and marine and coastal scientists.

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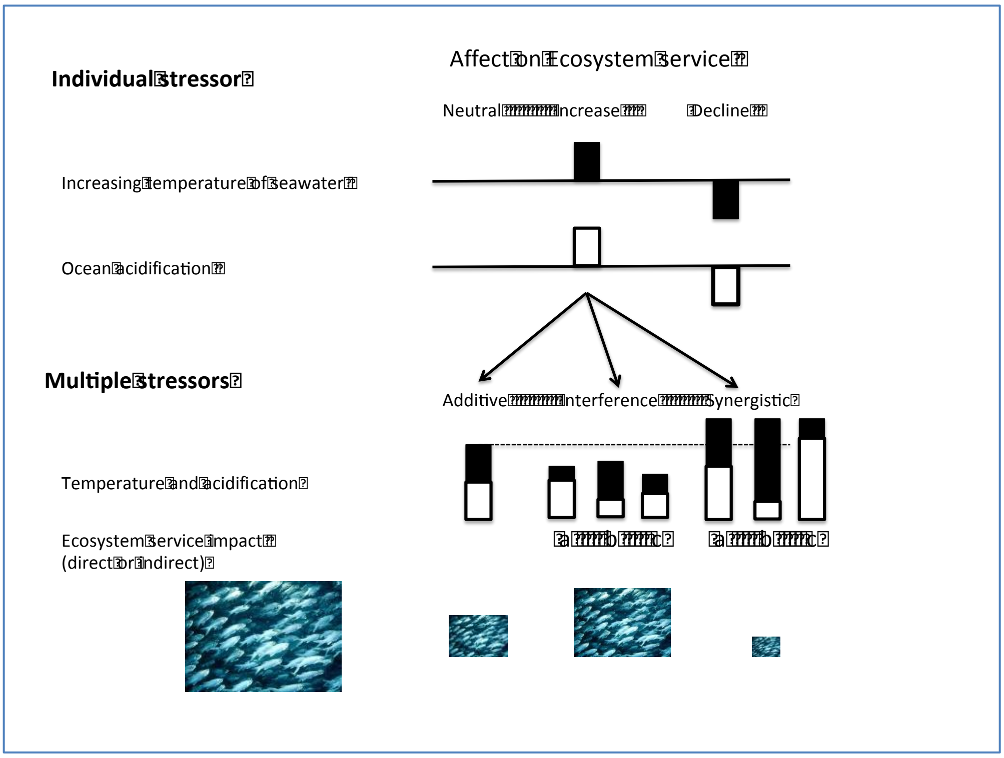

Figure 1. Stressors and affect on ecosystem services. An individual stressor may have a predictable impact on a selected ecosystem service (neutral, increase or reduce the service). Two stressors (multiple stressors) both impacting the same system may interact in terms of the ecosystem service outcome: being additive; interfering with each other; or behaving synergistically. It is this combined effect that will impact ecosystem service flows such as fisheries catch.

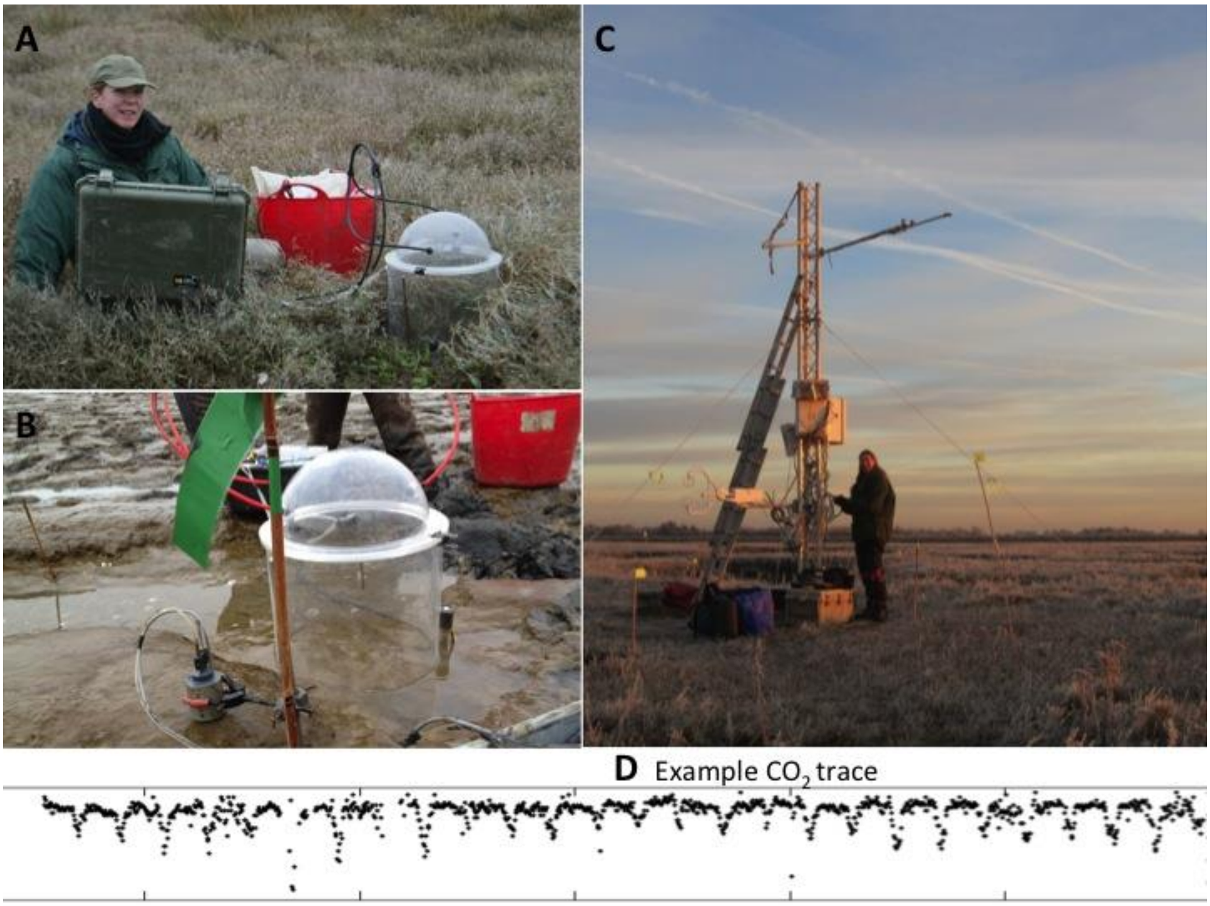

Figure 2: Research during the NERC CBESS campaign to assess carbon flows from and over coastal salt marshes in the UK. A: and B: The use of benthic flux chambers in situ. C: Large scale measures over the march using eddy covariance towers. Courtesy Dr Melanie Chocholek

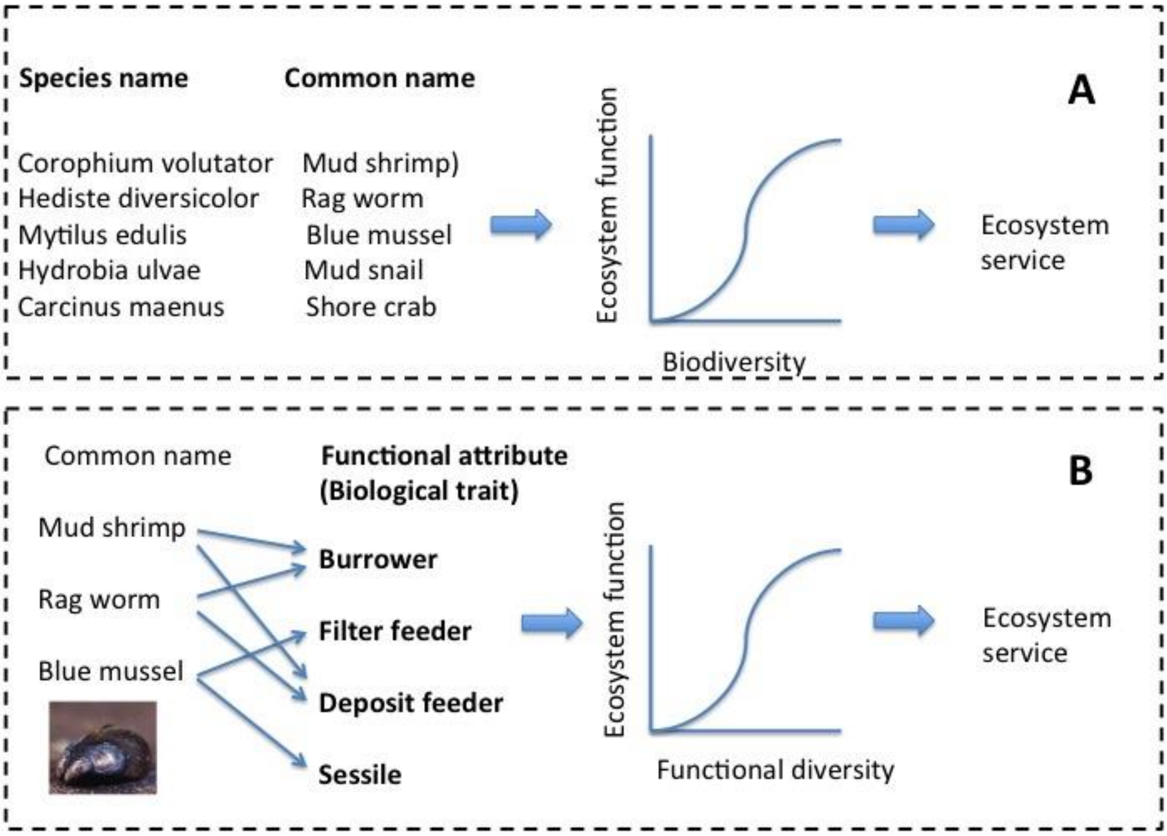

Figure 3. A. The theoretical link between ecosystem function and biodiversity describing an increase in functionality with increasing diversity until a maximum level of performance is achieved and the addition of further species has no more effect. B. Where species diversity is replaced by the traits of the species in the assemblage suggested as a more direct methods of linking species activity to ecosystem function.

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)

Table 2. Descriptors of Good Environmental Status under the Marine Strategy Framework Directive.

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 which are 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 and marine environment.
Non-indigenous species introduced by human activities have minimal affect on native ecosystems.
Sea-floor integrity is at a level that ensures that the structure and functions of the 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.

Source: HM Government, 2012.