Integrating methods for ecosystem service assessment and valuation: Mixed methods or mixed messages?

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Abstract

A mixed-method approach was used to assess and value the ecosystem services derived from the Dogger Bank, an extensive shallow sandbank in the southern North Sea. Three parallel studies were undertaken that 1) identified and quantified, where possible, how indicators for ecosystem service provision may change according to two future scenarios, 2) assessed members of the public's willingness-to-pay for improvements to a small number of ecosystem services as a consequence of a hypothetical management plan, and 3) facilitated a process of deliberation that allowed members of the public to explore the uses of the Dogger Bank and the conflicts and dilemmas involved in its management. Each of these studies was designed to answer different and specific research questions and therefore contributes different insights about the ecosystem services delivered by the Dogger Bank. This paper explores what can be gained by bringing these findings together post hoc and the extent to which the different methods are complementary. Findings suggest that mixed-method research brings more understanding than can be gained from the individual approaches alone. Nevertheless, the choice of methods used and how these methods are implemented strongly affects the results obtained.

1. Introduction

The concept of ecosystem services, the contributions of ecosystems to human well-being (de Groot et al., 2010), is a useful approach for demonstrating the links between humans and the environment. It is readily acknowledged that many of these services go unrecognised (or under-recognised) in the environmental management process (Daily, 1997; Dasgupta et al., 2000). Cumulative impacts and trade-offs between them are overlooked (Phal-Wostl, 2007; Lester et al., 2010). This often occurs because they may be used indirectly, or enjoyed directly (but maybe unconsciously), but are not traded through markets (Böhnke-Henrichs et al., 2013). It may also result because the links between environment and human well-being are indirect, occurring at different spatial and temporal scales (Corvalan et al., 2005). To overcome this problem, quantification and valuation of ecosystem services has been advocated as a solution (e.g. Liu et al., 2010). Valuation can be approached from multiple perspectives, including ecological value (the degree to which an ecosystem component contributes to an objective or condition such as an ecosystem service; Farber et al., 2002), economic value (often expressed in monetary terms; Brown, 1984) and socio-cultural value (or shared social values obtained through social interaction, open dialogue and social learning; Stagl, 2004).

Through assessment and valuation, the link between ecosystem services and human well-being is made more explicit (Fisher et al., 2009). Evidence of this link should therefore improve environmental decision-making, ensuring valued ecosystems continue to deliver the services essential to human well-being (Daily et al., 2009). Thus far, the many challenges involved in ecosystem service assessments and valuations have limited their use (Laurans et al., 2013), but within many environmental management circles, including marine planning, there is a growing call for wider ecosystem service assessment and valuation (e.g. Mooney et al., 2005; Börger et al., 2014a).

1.1 Quantification of Ecosystem Services Through Ecological Assessment

Interest in ecosystem service quantification has led to numerous ecological assessments of ecosystem services. These typically identify indicators of ecosystem services, attempt their quantification and spatial mapping (e.g. Burkhard et al., 2012; Crossman et al., 2013) and demonstrate how they have changed over time and/or model how they may change into the future (e.g. Martín-López et al., 2010). For marine and especially offshore ecosystems, no examples known to the authors exist that involve all these steps and apply them to multiple ecosystem services. Such assessments, however, may be particularly useful for ecosystem management because they facilitate the analysis of ecosystem service trade-offs made between alternative management options or possible future scenarios.

Being based on suitable indicators, outcomes of ecological assessments reflect ecosystem change (Hattam et al., 2015). They demonstrate the ecological importance of the system and can also assist with identifying the processes involved in ecosystem service supply (e.g. Cook et al., 2014). This facilitates the identification of drivers of change, which can also inform ecosystem management. Ecological assessments allow the investigation of a broad range of ecosystem services based on existing data. Hence they help identify and quantify the most important ecosystem services and those most intensely affected by human activities in an area. It is important to note that while ecological assessments explore how the supply of ecosystem services change over time, they do not provide information about the value of these ecosystem services to society. By quantifying expected changes they can, however, inform the development and application of valuation studies that explicitly aim to assess the social and economic value of the benefits derived from ecosystem services. In an attempt to encourage ecological assessments of ecosystem services, guidelines for doing this have been produced by organisations and institutions (e.g. IPIECA, 2011; EU, 2014).

1.2 Economic Valuation of Ecosystem Services

Economic valuation of the benefits from ecosystems is commonly the next step in the assessment (Defra, 2007). Economic valuation provides a common currency for units of value. This, it is argued, provides a means for comparing the costs of environmental protection with the benefits generated, and for comparing different management or policy goals, including environmental protection (Balmford et al., 2002; Hanley and Barbier, 2009). A further justification is that it should encourage more sustainable use of the environment and better motivate its conservation and protection (Daily and Matson, 2008; Tallis et al., 2008). Public bodies are increasingly offering guidance to environmental managers on how to undertake such valuations (e.g. HM Treasury, 2003; Pearce et al., 2006; Defra, 2007; Hansjürgens et al., 2012; Baker and Ruting, 2014) and incorporate the findings into policy and practice (e.g. Defra, 2010).

The value of ecosystem service benefits that are not traded in markets can be assessed using non-market valuation techniques (Cooper et al., 2013). Borrowing the logic of voluntary exchange in the market, such assessments typically aim to gauge people's willingness to trade some fraction of their wealth or income for an increase in ecosystem service provision. This willingness-to-pay (WTP) is interpreted as an indicator of the change in utility the person expects from the consumption of these increased ecosystem services. When WTP cannot be assessed through market data, survey-based techniques, such as the contingent valuation method (CVM) (Carson and Hanemann, 2005) and discrete choice experiments (DCE) (Hanley et al., 1998, Louviere et al., 2000) can be employed. These methods elicit WTP in a hypothetical market setting created in the survey interview. In the marine environment, the majority of valuation studies have been applied to coastal and near-shore ecosystems (e.g. Ressurreição et al., 2012; Hynes et al., 2013; Loomis and Santiago, 2013), but a growing number of applications to offshore and deep-sea sites and fauna can be found (e.g. McVittie and Moran, 2010; Wattage et al., 2011; Jobstvogt et al., 2014; Aanesen et al., 2015).

1.3 Alternatives to Economic Valuation

Economic valuation interprets private households as consumers of ecosystem services rather than as citizens holding attitudes and values regarding the provision of ecosystem services for society (Blamey et al., 1995; Orr, 2007). Consequently, this framework has been criticised from both within the field of economics (e.g. Aldred 2006; Parks and Gowdy, 2013) and elsewhere (e.g. Adams, 2014). Economic valuation techniques such as survey-based elicitation of WTP and concepts such as ecosystem services and natural capital frame the nature-society relationship into one of utility and exchange prefiguring commodification as a reasonable response (Kallis et al., 2013). Gómez-Baggethun et al. (2010) argue that even though the focus on economic valuation and payment schemes has attracted political support for conservation, it has also led to the commodification of a growing number of ecosystem services and the reproduction of the neoclassical economics paradigm and market logic to tackle environmental problems. There are competing values and interests relating to the environment between different groups and communities, something that also creates conflict among the groups and among communities across space and time (Martinez-Alier et al., 1998). Kosoy and Corbera (2010) highlight three invisibilities in the commodification of ecosystem services: (i) the technical difficulties and ethical implications that exist when narrowing down the complexity of ecosystems to a service or range of services, and how that changes the way we relate to and perceive nature; (ii) the fact that commodification of ecosystem services requires a single exchange-value, which in turn denies the multiplicity of values attributed to these services (i.e. there are values beyond monetary values that are important); and (iii) the fact that it reproduces rather than addresses existing inequalities in the access to natural resources and services.

Non-monetary approaches such as deliberative group discussions (Wilson and Howarth, 2002), citizens' juries (Spash, 2007) and q-methodology (Pike et al., 2014) utilise group based activities and participatory and deliberative approaches to attain detailed information about people's relationship with the natural environment and the socio-cultural values they place on it (Christie et al., 2012). Deliberation can refer to two kinds of discussions: one that involves a group of people who through deliberation carefully weigh reasons for and against some proposition, and another that involves an interior process by which an individual weighs reasons for and against courses of action (Fearon, 1998). Unlike conventional non-market valuation techniques such as CVM or DCE, which attempt to elicit pre-existing preferences or those constructed at the time of the interview, deliberative group methods, including citizens' juries, are based on the assumption that the values people hold regarding matters of collective choice can be constructed through the process of reasoned discourse with other members of society (Wilson and Howarth, 2002; Howarth and Wilson, 2006; Spash, 2007). In recognition of this, and the criticisms against economic valuation, public bodies are also providing guidance on a range of deliberative methods for the assessment of ecosystem services (e.g. Fish et al., 2011).

1.4 Integrating Methods

Despite calls for the integration of methods that elicit ecological, socio-cultural and economic values (e.g. de Groot et al., 2010; Lopes and Videira, 2013), most ecosystem service assessments focus on just one of these approaches, or combine ecological assessments with some form of economic or non-monetary valuation (e.g. Pascual et al., 2011; Pascual et al., 2012). In some cases mixed methods are applied drawing on both economic and non-monetary techniques (e.g. Szabó, 2011; Kenter et al., 2013). What rarely happens is a synthesis of the findings arising from the different approaches. Only two published papers have been identified within this study that attempt to integrate the outputs from biophysical, socio-cultural and economic approaches using empirical data (Castro et al., 2014; Martín-Lopez et al., 2014).Research into mixed-methods, however, indicates that multi-strategy approaches to research can bring more understanding than can be gained from the individual approaches alone (Bryman, 2006). Effort is therefore needed to understand how the different approaches to ecosystem service assessment and valuation support each other, or not, as the case may be.

Using the Dogger Bank (a shallow sandbank in the southern North Sea) as a case study, this paper explores the complementarities between three approaches to ecosystem service assessment and valuation: 1) an ecological assessment, which identified and quantified, where possible, indicators for ecosystem services delivered by the Dogger Bank and explored how these services may change according to two future scenarios, 2) a DCE, which assessed members of the UK public's WTP for improvements to a small number of ecosystem services provided by the Dogger Bank as a consequence of hypothetical management plans, and 3) a citizens' jury workshop that allowed members of the UK public to explore the uses of the Dogger Bank and the conflicts and dilemmas involved in its management. Complementarity analysis is just one approach to combining mixed method data (see e.g. Brannen, 2005), but is particularly suitable for data that have been collected through different methods at the same time (Teddlie and Tashakkori, 2009). The exploration of complementarities between these methods was undertaken retrospectively and was not planned as part of the original study. The approach taken is therefore only an example of how a synthesis stage could be undertaken. Ideally, integration should be planned from the outset with full understanding of what is required of the integrating approach. The growing call for evidence-based policy and practice combined with limited opportunities for primary data collection, suggests that such retrospective synthesis of data pertinent to ecosystem service assessments and valuation may become increasingly relevant.

By exploring the complementarities between the approaches used in this study, this paper "seeks elaboration, enhancement, illustration, clarification of the results from one method with the results from another" (Greene et al., 1989, p. 259). It therefore addresses the following research questions: To what extent do the different approaches used complement each other? How can the different methods be used more effectively together? And how can the findings be better incorporated into environmental management?

The paper is structured as follows. Section 2 introduces the Dogger Bank before providing a brief description of the methods used in each sub-study and the approach used to explore the complementarities between these methods. This is followed in Section 3 by a presentation of the results. The findings are then discussed in Section 4, with conclusions provided in Section 5.

2. Case Study and Methods

2.1 The Dogger Bank

Covering an area of 18,700 km^2, the submerged sandbank of the Dogger Bank is located in the southern part of the North Sea (Fig. 1). It is an important location for commercial fishing as well as actual and potential energy generation. The UK Government is planning the world's largest offshore wind farm to be installed on its section of the Dogger Bank (Forewind, 2010). It also provides a number of other less recognised benefits, for example, it acts as a nursery ground for fish (Diesing et al., 2009; Hufnagl et al., 2013) and it makes a contribution to carbon storage and sequestration, which in turn supports the regulation of the climate. In addition it is of cultural importance: fishermen and archaeologists have found a number of prehistoric remains on the Dogger Bank, and a small number of recreational anglers and scuba divers visit the Dogger Bank every year. As a consequence of its ecological importance and its vulnerability to human pressures, the UK, Germany and The Netherlands have designated their parts of the Dogger Bank as a Special Area of Conservation (SAC) under the EU Habitats and Species Directive (92/43/EEC) for the protection of Annex I Habitat H1110 ‘sandbanks which are slightly covered by seawater all the time’ (EC, 1992). This designation requires that all human activities within the SAC are regulated to fulfil the conservation objectives for the site.

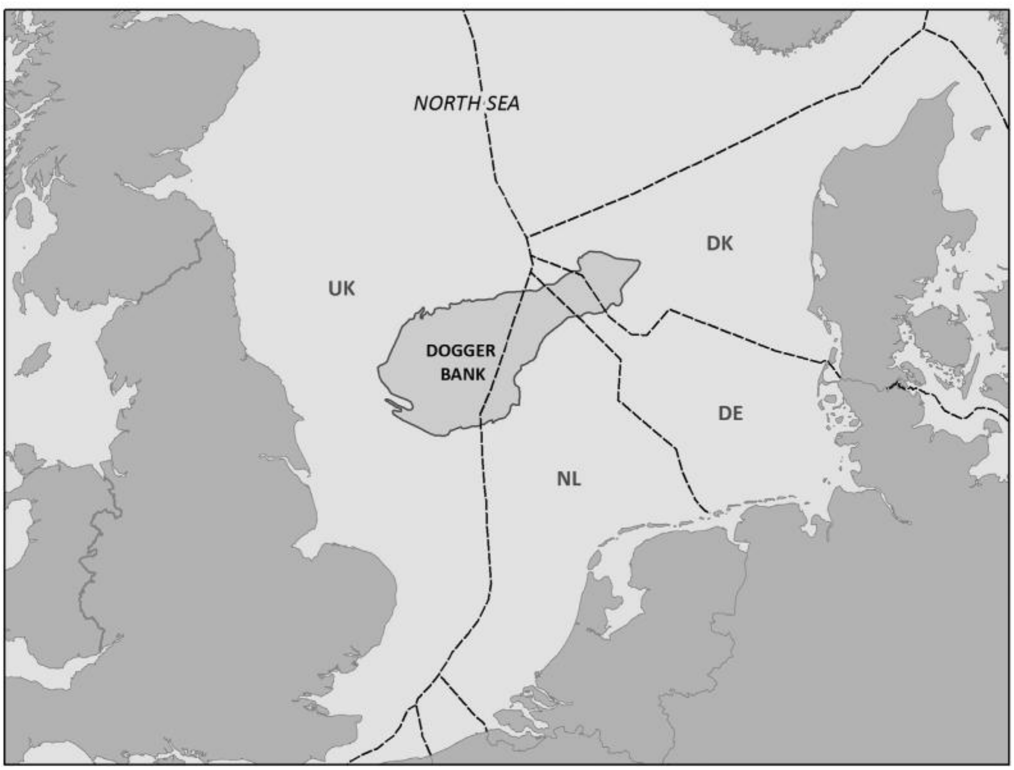

Fig. 1. Location of the Dogger Bank (UK - United Kingdom; DK — Denmark; DE — Germany; NL — Netherlands).

Management measures are currently under negotiation between the UK, Germany and The Netherlands before submission to the EU. Proposals for these management measures formed the backdrop to the DCE and citizens' jury scenarios.

2.1. Methods Applied

The ecosystem service framework and indicators defined by Hattam et al. (2015) formed the basis for this study. The three assessment and valuation studies then proceeded in parallel. The exploration of complementarities was undertaken post hoc and was not originally foreseen during the study development and planning phase.

2.1.1. Ecological Assessment

The main aim of the ecological assessment was to explore which ecosystem services are subject to change under different future scenarios. Indicators of ecosystem service quantity and quality were developed for all ecosystem services identified as relevant for the Dogger Bank (for details see Hattam et al., 2015). For clarity and to facilitate the assessment, indicators of ecosystem services (i.e. of ecosystem service supply) are considered distinct to indicators of ecosystem benefits (i.e. the outputs of ecosystem services, created and derived by humans). Attempts were made to quantify each of the indicators identified. The absence of appropriate data meant that indicators for only six of the ecosystem services identified could be assessed (Table 1).

Table 1
Ecosystem services and their indicators as assessed for the Dogger Bank.

Ecosystem services	Dogger Bank specific indicators	Measurement (units) – measured over time
Food provision – wild capture sea food	Population of nephrops, cod, haddock and flatfish species such as plaice, turbot and lemon sole	Biomass (tonnes km^{-2}) of fish and shellfish
	Quality of the populations of nephrops, cod, haddock and flatfish species such as plaice, turbot and lemon sole	Species composition, age profile; length profile; % affected by disease; mortality rates
Biotic raw material	Population of sandeels	Same measurement units as for food provision
	Quality of the populations of sandeels	Same measurement units as for food provision
Climate regulation	Air-sea and sediment-water fluxes of carbon and CO_2, scaled to the area covered by the Dogger Bank	Modelled (mg C m^{-2} d^{-1})
	Levels of carbon in different components of the marine ecosystem, scaled to the area covered by the Dogger Bank	Modelled carbon levels: biomass of carbon (g m^{-2}); dissolved organic or inorganic carbon (mg C m^{-3}); suspended organic or inorganic carbon (mg C m^{-3}); buried particulate organic or inorganic carbon (mg C m^{-2})
	Permanence of carbon sequestration, scaled to the area covered by the Dogger Bank	% of annual carbon turnover from sediments
	Air-sea fluxes of other greenhouse gases (e.g. dimethyl sulphide, methane, nitrous oxide), scaled to the area covered by the Dogger Bank	Examined, but neither modelled nor empirically determined (μg greenhouse gases m^{-2} d^{-1}) data available
Migratory and nursery habitat	Spawning: abundance of cod, sandeels, plaice, nephrops Nursery: abundance of sprat, nephrops	Abundance m^{-2} and species diversity
Gene pool protection	Diversity of species and sub-species, phylogenetic distance, Biodiversity Intactness Index	Expert judgement on species change and changes to Biodiversity Intactness Index
Leisure, recreation and tourism	Species of recreational interest e.g. harbour porpoise, grey seal, seabirds, fish	Count data of key species of recreational interest
	Area of biotopes of key interest to recreational users, scaled to the area covered by the Dogger Bank	Expert judgement on changes in area of biotopes of key interest to recreational users

To evaluate how the services provided by the case study sites may change in the future, present day (2000–2009) provision was assessed and compared against intermediate future provision (2040–2049). Two contrasting scenarios were used based on the IPCC 2002 National Enterprise (A2) and Global Community (B1) scenarios (IPCC 2000), a description can be found in Groeneveld et al. submitted and at http:// www.marine-vectors.eu/Core_pages/Future_scenarios_and_policy_implications_with_rele). Briefly, both scenarios encompass intermediate levels of economic growth but A2 envisages modest local environmental policy and limited global environmental policy, whilst B1 has ambitious local and global environmental policy. These global scenarios were augmented with location specific information (e.g. the B1 scenario included the construction of the existing planned wind farm on the UK sector and related fishing restrictions). Ecosystem service indicators were then assessed using various types of data, including measured data (e.g. fish catch data), modelled data (POLCOMS-ERSEM model output; Artioli et al., 2014) and data reported in the literature. Additionally, expert judgment was used to qualitatively identify possible effects of the scenario on ecosystem service provision. See Hattam et al. (2014) for more detail.

2.1.2. Discrete Choice Experiment (DCE)

In the absence of market data for the majority of ecosystem services provided by the Dogger Bank, primary valuation data were also collected through a survey with members of the public (Börger et al., 2014b). The survey used a DCE (Hanley et al., 1998; Louviere et al., 2000) to elicit the WTP of members of the UK public for securing some future positive environmental change (or to prevent some negative change from happening) on the Dogger Bank. As far as possible, the attributes of the DCE were linked to the ecosystem service indicators developed for the ecological assessment and targeted towards indicators for which no quantitative data were available.

The survey was undertaken online during December 2013. It presented respondents with hypothetical management measures drawn from the negotiations held by the Dogger Bank Steering Group about proposed fisheries management plans for the Dogger Bank (NSRAC, 2012). Respondents were informed that management would regulate fisheries and wind farm development (JNCC, 2011) and that these regulations would affect different aspects, or attributes, of the ecosystem: overall species diversity; the protection of seals, porpoises and seabirds; and the spread of invasive species. Respondents were asked to choose between the current, no cost situation and different management scenarios, each with differing impacts on the ecosystem attributes and associated implementation costs (Table 2). The inclusion of the cost component means that the value respondents attach to the different attributes can be inferred from respondents' stated choices and expressed as marginal WTP. For further details see Börger et al. (2014b).

Table 2
Choice attributes (current, no costs situation in italics).

Attribute	Description in the questionnaire	Levels
Diversity of species	Reducing or removing trawling in some parts of the Dogger Bank will: •Increase the diversity of fish, invertebrates and other marine species •Enhance the natural functions provided by the Dogger Bank (contributing to the regulation of climate, maintenance of clean water and support of fish populations)	*No change*, 10% increase in species diversity, 25% increase in species diversity
Protection of porpoises, seals and seabirds	The Dogger Bank provides a natural home for porpoises and seals, and is a feeding ground for seabirds. •These animals and birds are sometimes accidentally caught in fishing nets. •The use of harmful nets will be regulated or forbidden on some parts of the Dogger Bank meaning these animals will be better protected. •Fishing vessels will not be banned from the whole area.	*Not protected*, protected on 25% of the Dogger Bank area, protected on 50% of the Dogger Bank area
Invasive species	The construction of wind turbines on the Dogger Bank provides space for invasive species, increasing their ability to spread elsewhere. •They may affect the survival of species normally found there. •The higher the numbers of turbines and the closer they are, the greater the likelihood of invasive species becoming established.	*Restricted spread*, wide spread
Additional tax	Monitoring and enforcing the Dogger Bank management plan will be costly. The government therefore needs to raise additional funds through taxes. •The tax is payable by all households in the UK for the next 5 years. •If the overall funds people are willing to contribute do not cover the cost of monitoring and enforcement, the plan cannot be put into action.	*£0*, £5, £10, £20, £30, £40, £60

2.1.3. Citizens' Jury

As an alternative to economic valuation, a citizens' jury workshop on the Dogger Bank was held in Newcastle, UK, in October 2013 with 19 members of the UK public. Participants were selected from the database of a marketing company, according to particular criteria (e.g. age, gender, socio-demographic status). It was anticipated that there would be a lack of knowledge among workshop participants about the Dogger Bank, and hence background information would need to be provided to facilitate discussions. Accordingly, the workshop was based on the principles of a citizens' jury in which expert witnesses are invited to state their case to a group of jurors selected from the general public (Huitema et al., 2007). Expert witnesses are people who are knowledgeable of the issue in question or strong advocates of particular positions in the debate. After hearing all the witnesses' accounts, the jurors (the participants) deliberate together on the issue in attempt to reach a common 'verdict' or conclusion. As consensus-seeking processes may silence minority perspectives (Travers, 1990), the primary aim of the Dogger Bank workshop was not to get participants to arrive at a common conclusion. Instead, it aimed to understand all the diverging perspectives and positions, arguments, nuances and stakes which are represented among the participants, as well as how the group setting influenced the formation of opinions. It therefore explored shared social values, focusing on aspects of use and non-use of the Dogger Bank.

Participants were provided with information from expert witnesses about the Dogger Bank environment, the uses of the Dogger Bank and their impacts on the marine environment. Witnesses included representatives of the fishing and wind energy sectors, a marine biologist and a speaker putting forward the position of environmental nongovernmental organisations (ENGOs) involved with discussions on the Dogger Bank management plan. After hearing the witness presentations, participants were divided into four groups for two rounds of facilitated discussion. The first round focused on “what does the ocean mean to you?”, “what should we use the ocean for?”, and “uses of the Dogger Bank and the implications of this use”. The second session focused on “conflicts and dilemmas in the management of the Dogger Bank” and “ranking competing uses of the Dogger Bank”. Throughout the workshop, participants were reminded that the word ‘use’ was meant to cover all things provided by the ocean and the Dogger Bank that respondents and society might find of value or meaningful. This avoided the need to use the term ecosystem services and the discussion of the meaning of ecosystem services that might result. More information about the workshop can be found in Hattam et al. (2014).

2.1.4. Exploration of Complementarities

The synthesis of the findings from the above methods was undertaken once the results were available from each stage. The three methods described were applied concurrently, which allowed for a parallel track analysis (Teddlie and Tashakkori, 2009). Parallel track analyses are particularly suited to exploring complementarities as the data are analysed at the same time and the findings emerge together. This is the most common mixed analysis technique and “although the ... sets of analyses are independent, each provides an understanding of the phenomenon under investigation. These understandings are linked, combined, or integrated into meta-inferences” (Teddlie and Tashakkori, 2009, p. 266).

The first stage in the assessment was to explore the complementarities between methods themselves and the way they were applied, rather than between the outcomes of those methods. This involved examining the complementarities between the work steps taken in the application of the methods, followed by a matrix cross-tabulation, in which each method was compared against a set of criteria. Criteria ranged from what is being valued and how the value is expressed, to the types of data used, the approach to data analysis and interpretation, the transferability of related outcomes and the strengths and weaknesses of the methods. The second stage focused on the complementarities between the results. This drew loosely on Greene (2007) and involved data transformation, whereby the quantitative findings from the DCE were expressed as a narrative to facilitate the comparison of mixed data types. Using matrix cross-tabulation, the relationships between findings were examined. This focused on the convergences and divergences between the findings and the trade-offs for management implied by them. The final stage in the assessment involved the drawing of inferences and conclusions. This approach provides just one example of how to explore the complementarities and combine the outputs of different methods.

3. Results

This section presents summary results for each method used. It emphasises the types of results obtained and key findings only. Full details on how these results were derived can be found in Hattam et al. (2014) and Börger et al. (2014b).

3.1. Ecological Assessment

Ecological indicators for this assessment were selected according to those that would best reflect the quantity and quality of the ecosystem service provision. To quantify these indicators, ecological assessments of ecosystem services as performed in this study require data relating to both the functioning of ecosystems, as well as quantifying what species or habitats are present or absent. While ecological data are available for the Dogger Bank, they are largely unsuitable for such assessments being either insufficiently resolved spatially, incomplete, or poorly resolved and understood in that area. If indicators could not be quantified, they were not replaced with inferior indicators, the services were simply left unassessed. Limitations in data availability and knowledge therefore restricted the possibilities for the ecological assessment of ecosystem services based on secondary data.

Quantitative data were available to assess the current state of 20 indicators corresponding to six ecosystem services. Modelled future projections, however, were only available for the indicators of climate regulation (Butenschön and Kay, 2013). Assessments of change are therefore primarily based on the expert judgment of the multidisciplinary authors and mainly serve as an example of how changes in ecosystem services may be measured. The main output of this assessment is a qualitative statement of change (Table 3) for each of the ecosystem service indicators listed in Table 1. Information obtained from these indicators represents only a partial account of the situation found on the Dogger Bank. Where the assessment was based on expert judgment, or where indicators were insufficiently supported by data for any kind of assessment, the results highlight data gaps and areas for future study.

As might be expected, the B1 (Global Community) scenario presents a much more positive future than A2 (National Enterprise) in terms of ecosystem service delivery (Table 3). Under the B1 scenario most indicators are anticipated to show upward trends or no change from the present. The downward trend for the fishery mortality indicator (see sea food and raw materials) requires care in its interpretation as it actually translates into positive overall change for fish stocks. Under A2 most indicators show downward trends or no change, suggesting that the related ecosystem services are decreasing. While useful in intimating future trends in ecosystem service supply, this assessment does not support the drawing of conclusions about changes in the relative values or importance of individual ecosystem services.

Table 3
Future trends in ecosystem service provision from the Dogger Bank area under two alternative scenarios. Indicators in italics have been assessed using modelled data, assessments of change in all other indicators are based on expert opinion.

			Dogger Bank	
Ecosystem services	High level indicator	Specific indicator	A2 scenario	B1 scenario
Food provision - wild capture seafood	Fish/shellfish populations	Biomass	↓	↑
		Abundance	↓	↑
	Quality of the fishery	Species composition	↓	↔
		Age profile	↓	↑
		Length profile	↓	↑
		Fishing mortality	↑	↓
		% affected by disease	↔	↔
Biotic raw materials	Quantity of raw materials	Biomass	↓	↑
	Quality of raw materials	Mortality	↑	↓
Climate regulation	Air–sea and sediment– water fluxes of carbon and CO_2	*Air–sea flux*	↑	↑
		Carbon burial	↔	↔
		Total organic carbon	↓	↑
	Air–sea and sediment– water fluxes of other greenhouse gases	*Air–sea flux*	?	?
Gene pool protection	Genetic diversity	Species diversity	↔	↑
		Biodiversity intactness index	↓	↑
Nursery and migratory habitat	Number and diversity of species using the area for nursery or reproduction	Abundance of fish/shellfish eggs	↓	↑
		Abundance of fish/shellfish larvae	↓	↑
	Dependence of off-site (commercial) populations	Dependence of off-site commercial species	↔	↔
	Area of habitat or density of biogenic habitat creating species "used" or identified as important for nursery or reproduction	Area of biogenic habitat	N/A	N/A
Leisure, recreation and tourism	Species of recreational interest	Seals, cetaceans and birds	↓	↑(but opposite for birds)
	Biotopes of recreational interest		↔	↓

3.2. Discrete Choice Experiment (DCE)

Four types of results were produced from the DCE (Börger et al., 2014b):

1. Coefficients from choice models, which indicate the effect of attributes on choices;
2. WTP estimates as an expression of value and as an indicator of expected utility change resulting from the ecosystem changes described in the choice attributes (Table 4);
3. Respondent-specific determinants of different coefficient patterns (and thus WTP estimates) allowing differentiation between groups of respondents who hold different preferences; and
4. Measures of unobserved, i.e. random, heterogeneity of preferences across respondents.

Results show that the respondents hold significant values for environmental benefits generated by the proposed management measures. Ecosystem attributes positively affect choice (i.e. the probability that a management option is chosen over the business-as-usual option), while cost negatively affects choice. These respective influences increase with the level of the attribute/cost. WTP for the protection of porpoises, seals and seabirds was higher than for restricting the spread of invasive species and general species diversity respectively. This implies that restrictions to fishing using nets that protect these charismatic species are preferred to restrictions to fishing using bottom trawling techniques that protect species diversity in general, as explained by the management scenario that framed the choice tasks.

Table 4
Implicit prices of consequences of a hypothetical Dogger Bank management plan as elicited in the DCE survey (Börger et al., 2014b).

Attributes		Mean WTP (£)	95% confidence interval
Species diversity	- no change[a]		
	- 10% increase	4.19	[0.70–7.69]
	- 25% increase	7.76	[5.15–10.35]
Protection of charismatic species	- no protection[a]		
	- on 25% of Dogger Bank area	24.02	[20.66–27.38]
	- on 50% of Dogger Bank area	30.32	[27.02–33.62]
Invasive species	- restricted spread[a]		
	- wide spread	−25.39	[−28.51 - -22.28]

WTP was calculated from a random parameters logit model with 5000 Halton draws based on a sample of 973 respondents completing six choice tasks each. Confidence intervals were computed based on the bootstrapping approach by Krinsky and Robb (1986).
[a] Indicates the current, no cost situation.

Respondents who are members of an environmental organisation and have previously taken a ferry or flight over the North Sea prefer management measures for the Dogger Bank more often than respondents without these characteristics. Holding attitudes that favour the introduction of a management plan to protect species diversity and charismatic species also increases the WTP of respondents for different increases in the corresponding attributes. In addition, random preference heterogeneity is present that cannot be accounted for by respondent characteristics and attitudes. These findings show how DCEs can allow for some degree of diversity in values between respondents.

3.3. Citizens' Jury

Deliberations between respondents allowed multiple views on the ocean and the Dogger Bank to emerge. Participants were able to influence each other to generate new positions, with the shared experience affecting the outcomes. Responses to the questions "what does the ocean mean to you?" and "what should we use it for?" indicated the participants' views on the ocean as well as concerns over its use. Remarks such as "the integrity of the ocean", "importance of the function of the ecosystem", "the beauty of the natural environment", as well as use of words such as preservation, sustainability, protection and responsibility highlight the importance of the ocean beyond economic values. At the same time however, the importance of the economic uses of the ocean was embedded in participants' understanding, as the ocean was also viewed as a "human resource" and used for "getting the resource(s) [for humans]."

The key output of the citizens' jury workshop is an identification of discourses. A qualitative discourse analysis of these deliberations identified two main themes:

1. that fishing should be prioritised over wind farm development, and
2. that conservation should be a priority, but with specific caveats.

The prioritisation of fishing arose from what was considered to be a lack of evidence supporting the potential impacts or benefits arising from the construction of a wind farm on the Dogger Bank. It also arose out of the perceived historical legitimacy of fishing ("Fishing has been in place for years ... I don't feel that they are going to impact now because they have been there for so long.") and the ability of the expert fisheries witness to demonstrate the sustainability of the fishery on the Dogger Bank.

Conservation was a thread in many of the discussions with participants recognising the intrinsic value of the Dogger Bank. Conservation was not considered to exclude the use of the Dogger Bank for economic purposes, but ensuring this use is balanced and sustainable was highlighted by jury members. Many participants agreed that multiple activities should be allowed on the Dogger Bank through a system of zoning supporting both economic and non-economic uses. However, they felt that they lacked the information to discuss such zoning in more detail.

The deliberative exercises demonstrated the necessity for careful facilitation to ensure all views are heard and to understand the ways in which participants influence each other. For example, discussion uncovered that one of the participants worked in the energy management sector and was knowledgeable about renewable energy. This participant suggested convincingly during the question and answer session of the witnesses that offshore wind farms could lead to negative changes in biodiversity without reducing electricity bills. In the absence of data proving otherwise, this argument can be demonstrated to have influenced other participants' views on offshore wind farms.

3.4. Integration of Findings

In drawing together the three datasets, it is important to acknowledge the limitations of each. For example, the lack of quantitative data in the ecosystem service indicator assessment limits the understanding gained from their assessment. Consequently, the outputs largely reflect the direction of change indicated by the scenario narratives and the interpretation of the scenarios by the researchers. In the discrete choice experiment, the use of management measures to frame the choice experiment is novel, but makes interpretation of the results more challenging. It is not entirely clear whether respondents make choices on the basis of the management measure or the outcome of management (i.e. the attributes). The latter is more likely according to findings from a think aloud exercise conducted during the survey testing stage. For the citizens' jury workshop, more juries with different jurors and follow-up sessions with the same jurors would be needed to increase the level of confidence in the findings. It is possible that a jury with different jurors could have produced different results. Lastly, the size of the combined dataset is small, being based on only three studies. Had this integration been planned from the outset, the three methods may have been applied differently and additional or larger datasets sought. Despite these shortcomings, the potential to learn more from the combination of the data requires further attention. This will help to demonstrate the extent to which the data complement each other and whether a mixed methods approach can overcome any of the weaknesses in the individual methods.

3.4.1. Complementarities in Work-flow

Fig. 2 demonstrates how the workflow for the different methods overlapped and where the development of methods supported each other. The ecological assessments were particularly important in terms of framing the DCE and focusing the citizens' jury, at both the preparatory and final stages. The preparatory stages of the DCE and the citizens' jury were also complementary. Both methods drew on the same exploratory semi-structured interviews with members of the public that were used to set the scene. As anticipated, there was little flow from the DCE and citizens' jury back to the ecological assessment, except during the development of scenarios.

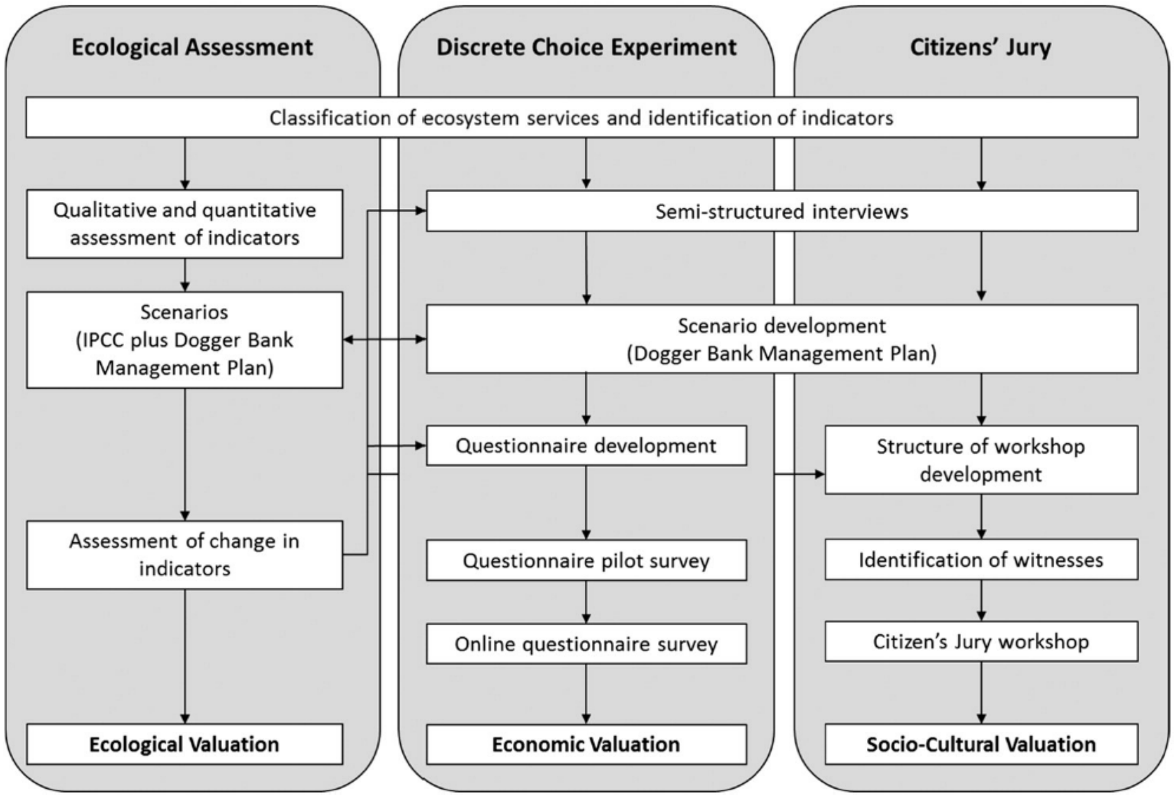

Fig. 2. Complementarities in work-flow between methods.

3.4.2 Complementarities Between Methods

In terms of methodological detail, a matrix was developed (Table 5) to facilitate comparison across the three methods applied in this study. The three method approach has allowed different value types for ecosystem services to be estimated supporting, to some extent, an assessment of both the supply of ecosystem services (via the ecological assessment) and the demand for some of these services (through the DCE and the citizens' jury). None of the methods used capture all aspects of ecosystem service supply or demand, however. Despite drawing across a diverse range of information sources, as has been found elsewhere (e.g. Liquete et al., 2013) there is a bias towards services for which more data and understanding exist (e.g. food provision and carbon sequestration). The bundling of services within the DCE and citizen's jury also means the findings are hard to interpret in terms of individual ecosystem services. The outcome of the three approaches and their integration is therefore a partial understanding of the ecosystem services of the Dogger Bank and how they will change. Nevertheless, considering their transferability, the findings from the DCE and the indicators may be useful for similar assessments in other locations. The results of the DCE are drawn from a national survey and therefore could be used in benefit transfer, if applied to sites with comparable characteristics and facing similar management scenarios (Richardson et al., 2014). The indicators used in the ecological assessment could also be transferred, but tailoring to different locations would be necessary.

Table 5
Method comparison and complementarity (as undertaken in this study).

Assessment method	Ecological assessment	Discrete choice experiment	Citizens' jury
Value type	Ecological value.	Economic (non-use) value.	Social/cultural value.
What is being valued?	Supply of individual ecosystem services.	Management and management outcomes. Demand for benefits arising from bundles of ecosystem services.	Activities/uses. Demand for environmental outcomes.
Output/unit	Units of quantity (e.g. tonnes of fish landed or available to be landed; tonnes of carbon sequestered) Units of quality (e.g. fish mortality rates, age profile).	Monetary values.	Discourses/themes. Preference ranking.
Directly address ecosystem services?	Direct.	Direct and indirect. Bundle of ecosystem services.	Indirect. Bundle of ecosystem services.
Information sources	Literature[a], expert opinion, ecosystem models, secondary data[b].	Literature, expert opinion, preparatory interviews (with the public), survey data.	Literature, expert opinion (stakeholders), deliberation (with public).
Public engagement	No.	Yes.	Yes.
Transferability of results	Indicators may be transferred, but: • may need tailoring to specific site; • may respond differently in different sites.	Potential use of results in benefit transfer (when targeted to similar ecosystem type, management scenarios with defined ecosystem services).	Findings are specific to location/issue of interest.
Weaknesses/limitations	Some ecosystem services easier to quantify and assess than others, leading to bias in findings. Absence of appropriate data limits applicability, especially in the marine environment.	Limited understanding of why one attribute favoured over others. Focuses on limited number and bundled ecosystem services. Meaning of monetary values influenced by questionnaire design. Bundling of services limits understanding of trade-offs. Communicating ecosystem services is challenging.	Links to ecosystem services are weak. Influenced by: workshop design, witnesses and information provided, more knowledgeable participants. Communicating ecosystem services is challenging.
Strengths (overcoming weaknesses)	Focus on multiple ecosystem services, provided data/literature and experts are available.	Provide monetary estimates of ecosystem services value relevant to cost-benefit analysis.	Provides in depth understanding of theme/discourse emergence.
Method complementarity	Provides broad picture of ecosystem service change. Helps identify ecosystem services suitable for valuation. All ES considered equal. Combined with preference data, useful for exploration of mismatches between ecosystem services supply and demand.	Provides monetary value estimates for ecosystem services with no market value.	Captures detail of people's priorities not reflected in monetary valuation.

Consideration of the strengths of each of the methods helps identify where the methodological complementarities lie. The scope of the ecological assessment has the potential to be broad and can therefore offer a more rounded assessment of how ecosystems and the services they deliver may change as a result of human action or environmental variability. It thus provides insights on the capacity of an ecosystem to generate ecosystem services and it can also direct where it may be more useful to focus valuation studies. Both the DCE and the citizens' jury provide some understanding of society's demand for ecosystem services and how changes resulting from management actions may be valued. In the case of the DCE, these outputs generate information on the contribution of ecosystem services to human well-being and the hierarchy of preferences for ecosystem services. DCE outputs may also be used in cost-benefit analysis. The information obtained from the citizens' jury can augment these findings by providing greater understanding of why people hold the priorities that they do. Combining the three methods can be used to explore mismatches between ecosystem service supply and demand, and consequently identify any trade-off that may be necessary or preferable to make through environmental policy and ecosystem management.

3.4.2. Complementarities Between Results and the Trade-offs Implied

Identifying complementarities between the results of the three methods is challenging, given the limited size of the dataset and some of the limitations present in how the individual approaches were applied. Nevertheless, some complementarities between the findings are apparent, as are implied trade-offs (Table 6). Overlap between the three methods focuses on the impacts of management activities on fisheries, wind farm construction and conservation measures. The exploration of complementarities therefore concentrates on this overlap.

Table 6
Complementarities between results and indicated trade-offs.

	Topic of overlap		
	Conservation	Fisheries	Offshore Wind Farms
Ecological assessment	Scenario B1 with least human pressure better for ecosystem services.	Suggests limitations to fisheries most favourable to supply of all ecosystem services.	B1 scenario would see extensive offshore wind development on the Dogger Bank.
Discrete choice experiment	WTP for charismatic species and species diversity conservation. WTP for conservation of charismatic species greater than for species diversity.	Preference for net fishing restrictions over restrictions to bottom trawling.	Preferences for or against wind farms not directly assessed. Respondents WTP for responsible wind farm design that limits invasive species.
Citizens' jury	Conservation a priority, although with caveats.	Fisheries considered historically legitimate.	Fisheries preferred over wind farms.
Trade-offs?	No. General agreement.	Yes. Partial agreement.	Yes. Limited agreement.

In the context of conservation issues, preferences for the supply and demand for ecosystem services appear to move in the same direction. The DCE and citizens' jury both indicate preferences for conservation, especially of charismatic species. This in turn indicates a preference for the outcomes of the B1 (Global Community) scenario of the ecological assessment. Conservation measures on the Dogger Bank will in part be delivered through fisheries management (NSRAC, 2012) and here there is implied disagreement between the findings. The ecological assessment indicates that the closure of fisheries would be beneficial for ecosystem services supplied by the Dogger Bank (scenario B1). The DCE results, however, suggest that restrictions to net fishing would be preferred over restrictions to bottom trawling. This means that preferences for conservation of charismatic species would be met, but bottom trawling would continue to deliver fish but with no benefit to species diversity. In contrast, the outcomes of the citizens' jury suggest that, in terms of use of the Dogger Bank, fishing should be prioritised over other uses as a result of historical legitimacy.

In terms of wind farm construction the picture is less clear. The B1 scenario would see a substantial increase in the number of wind turbines constructed on the Dogger Bank (while the A2 scenario would only see some increase). While the acceptability of offshore wind farms was not assessed in the DCE, the relationship between offshore wind farms and fisheries has implications for the supply of fish. Fishing does not usually occur in wind farming areas, due to concerns over gear entanglement and infrastructure damage (Mackinson et al., 2006). Any increase in wind farm extent will therefore reduce fishing opportunities, in partial contradiction with the preferences expressed in the DCE results and complete contradiction with those from the citizens' jury.

Despite these apparent contradictions in findings, the methods do offer complementarities. Both DCE and the citizens' jury lend support to management aimed at achieving the B1 scenario of the ecological assessment and not the A2 scenario. Furthermore, they provide enhanced understanding of why this is the case. The DCE and the citizens' jury findings also largely agree, but the partial disagreement is illustrative of the complexity behind people's understanding of and demands for fisheries management. Where partial agreements or disagreements between findings occur, this indicates areas where trade-offs may arise when management decisions are taken. It highlights a mismatch between the supply and demand for ecosystem services in an area. The main trade-off implied by this work is in the context of fisheries restrictions and the interaction between fisheries and wind farms.

4. Discussion

Börger et al. (2014a) highlight a growing demand for wider assessment and valuation of marine ecosystem services in support of marine planning. For example, in the UK, ecosystem services have been identified as a priority research area by the Marine Management Organisation, the Government body responsible for marine planning (MMO, 2014). In addition, there is a move towards national assessments of ecosystem services through the Intergovernmental science-policy Platform on Biodiversity and Ecosystem Services (IPBES) in support of the Convention on Biodiversity and, for example, the European Unions' Biodiversity Strategy to 2020. Assessment of marine ecosystem services, however, often lags behind the assessment of terrestrial ecosystem services, hindered by inadequate knowledge and lack of data (Townsend et al., 2014). Applying a mixed-method approach may therefore provide useful insights by delivering a more comprehensive understanding.

4.1. Do the Methods Complement Each Other?

Three key areas of complementarities have been explored: between the work-stages of each method, between the methods themselves and between the findings. Complementarity between work stages is apparent, but this largely depends upon the communication within the multidisciplinary research team. In this case different aspects of the work did feed into each other, for example, sharing of preparatory semi-structured interviews between the DCE and citizens jury, the use of multidisciplinary teams to develop scenarios and ensure ecological content validity in the DCE and citizens jury.

In terms of methodological complementarity, the different stages of the assessment can be used to enhance each other. For example, the data gaps emerging from the ecological assessment were used to direct the DCE and citizens jury, and each method covers a different aspect of value and more or fewer ecosystem services. Despite limitations in data availability, the ecological assessment was the broadest in scope. In contrast, the DCE and citizens jury provided greater detail about more focused topics and particularly about demands for different ecosystem services or management outcomes. The ecological assessments help to identify how those demands might be met.

The findings from the Dogger Bank case study show complementarities between results. Conservation priorities were clearly demonstrated in the DCE and citizens jury. This supports management actions that would lead to the more conservation focused scenario (B1 Global Community), which suggests a more positive future for ecosystem services. Even where divergence between findings is apparent (i.e. in the case of fisheries priorities), complementarities are evident as the outcomes from the citizens jury improve understanding of why this divergence occurred. Potential mismatches between supply and demand for ecosystem services are highlighted, as are possible conflicts between management objectives desirable from an ecosystem perspective (e.g. fisheries closures) and those preferred by society (e.g. fish). The outcome is a more comprehensive understanding of the complex issues relating to the management of the Dogger Bank, which may better inform decision-making.

4.2. Does the Application of the Mixed-Methods Approach Overcome any of the Weaknesses of the Individual Methods?

The ecological assessment provides a general picture of how the Dogger Bank ecosystem may change. It reflects the capacity of the Dogger Bank to supply ecosystem services and identifies services worth exploring in valuation studies. The DCE elaborates upon this, through the provision of estimates of monetary value for little explored ecosystem services and those for which no secondary data exist. The citizens jury furthers this understanding through an in-depth exploration of people s values, providing some explanation of individuals priorities. The citizens jury also allows greater understanding of members of the public s preferences for ecosystem management of the Dogger Bank and can be used to infer societal demand for ecosystem services beyond their economic value. Only by applying the different methods do the trade-offs between the supply of ecosystem services and the different demands for ecosystem services become apparent.

4.3. Applying the Methods More Effectively: Lessons Learnt

The findings from the three distinct methods applied here suggest a mixture of messages. These raise a number of issues that need to be considered if greater integration of findings is to be achieved from similar studies in future. Lessons include the need to plan for integration; the need for better understanding of what integrating involves; the limitations of data availability; and the need to carefully consider the use of scenarios across the approaches.

4.3.1. Planning for Mixed Method Integration

Method integration requires planning from the outset. Greater complementarity could have been found with different method combinations (i.e. using other methods than those applied here or applying the same methods in different ways). For example, the citizens jury discussions could have been conducted differently with additional deliberative sessions or information from different witnesses provided to participants. Ecosystem services could have been focused on more explicitly to allow greater comparability to the DCE. In the DCE, ecosystem services could have been decoupled from the management scenarios and focused more clearly on the ecosystem service indicators used in the ecological assessment. The bundling of services in the DCE made the valuation outcomes harder to interpret and only indirectly addresses potential future changes in the provision of ecosystem services. To some extent context influenced design of both the DCE and the citizens jury. Respondents unfamiliarity with the Dogger Bank necessitated simplification, and consequently bundling, that may be unnecessary in more familiar settings. The design and focus of individual studies and any integrating stage therefore requires very careful co-planning to minimise unwanted divergence.

4.3.2. Understanding Data Integration

Understanding what is needed for data integration could also influence the way in which individual valuations are undertaken. For example, greater emphasis could be placed on quantitative rather than qualitative data collection, or different approaches to integration could be used. Complementarity mixed-methods studies are typically used to measure different as well as overlapping aspects of the same issue. Other approaches, such as triangulation, require that different methods are used to study the same issue (Green et al., 1989). In situations where additional numerical data are available, quantitative integration may be possible. Martín-López et al. (2014) draw on multiple quantitative data sources to which, once standardised, they apply principal component analysis to identify the relationships between biophysical, sociocultural and monetary values. Ecosystem service assessment and valuation researchers may be able to learn lessons from disciplines where application and integration of mixed-methods is more commonplace (e.g. Greene 2007; Teddlie and Tashakkori, 2009).

4.3.3. Impacts of Data Limitations

The availability of suitable data hindered all methods used in this study, but in particular the ecological assessment. This absence of data, especially prevalent in the marine environment, presents a difficulty for future assessments. It is recognised as one of the main challenges for the incorporation of ecosystem service assessments and valuation into marine planning (Börger et al., 2014a). The gaps identified here indicate where future monitoring effort is needed if ecosystem services are to be incorporated into marine management for the Dogger Bank.

The absence of appropriate information for the citizens jury also affected the ability of members of the public to discuss the uses and benefits of the Dogger Bank, and how the Dogger Bank should be managed. Despite providing participants with background information and experts to question, they still felt they had insufficient information to make informed decisions. Follow-up sessions are needed with the same participants to allow them to reflect on the information they have received and allow further discussion, as well as additional workshops with different participants (e.g. Abelson et al., 2003). This would enrich the data from the citizens jury and provide increased confidence in the results.

Improving the effectiveness of complementary studies requires not only improvement in the input data used in the different methods, but also increased generation of data from the application of different methods. Additional economic valuation, through DCE surveys or other methods, is needed to cover a wider range of ecosystem services. For example, Martinez-Lopez et al. (2014) draw on seven monetary valuation studies covering nine ecosystem services. This suggests opportunity for benefit transfer, however, benefit transfer may present challenges for integration, if the data are being used for a purpose that is different to that for which the data were originally collected.

Alternatively, the outcomes of complementarity studies such as this could be used to focus future ecosystem service assessments and valuations of the same study site. This would enable complementarities or divergences emerging from the first cycle to inform the next. For example, the preferences highlighted by DCE and the citizens' jury could be used to focus future ecological assessments and modelling efforts. Any divergences apparent between methods could form the focus of deliberations in a future study or inform economic valuations such as DCEs.

4.3.4. Mismatches Between Scenarios Used

Future scenarios were incorporated into each of the three methods used in this study. A mismatch is apparent, however, in the time-frames used. The ecological assessments considered changes to 2050, a relatively short time-frame for ecological change, while the DCE and the citizens' jury explored change in the near future (undefined in the citizens' jury and over the next five years for the DCE). This mismatch results from the very different time-frames suitable for the different approaches. While for ecological assessments a five year time frame is in most cases too short for any change to become apparent, a 50 year period is far too long for workshop or survey participants to be able to assess. Furthermore, preferences are unlikely to be stable over such a long period meaning resulting preference data may be too uncertain for use in long-term environmental management.

This mismatch is not necessarily a problem and is potentially a strength of mixed-method approaches. The implications of current actions needed to achieve future ecological outcomes and the tradeoffs they imply can be more easily evaluated through mixed-method approaches. In addition, if accompanied by biological/ecological monitoring and updated assessments of societal and individual preferences, management could be adapted to better achieve desired goals. This would ensure ecosystem management is responsive not only to environmental change but also to changing preferences or societal demand.

5. Conclusion: Better Supporting Marine Management

Growing use of the marine environment demands careful spatial planning (Douvere, 2008; Douvere and Ehler, 2009). The integration of findings from different ecosystem service assessment and valuation approaches can highlight complexities relating to management outcomes (e.g. for the Dogger Bank in relation to fishing) that would not become apparent using a single method approach. The combination of an ecological assessment (describing the supply of ecosystem services) with a DCE and a citizens' jury (that assess ecosystem service demand) identified areas where mismatches may occur between ecosystem service supply and demand in the future. This study has also highlighted potentially contentious issues (e.g. fisheries management) that will require careful consideration if societal demands are to be balanced with conservation needs.

There will always be trade-offs between improving approaches to ecosystem service assessments and having the resources to cover all relevant aspects of such assessments. Including an integration stage at the end of ecosystem service assessments may allow researchers and funders to obtain greater understanding from their data. It may therefore prove a powerful tool for supporting environmental management decisions. As shown in this case study, mixed methods approaches can (and probably most likely will) generate mixed messages. Where those mixed messages are understood as challenges or used to focus ecosystem management, the full potential of mixed methods approaches can be utilised, offering more than single method approaches can deliver.

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References

Aanesen, M., Armstrong, C., Czaijkowski, M., Falk-Petersen, J., Hanley, N., Navrud, S., 2015. Willingness to pay for unfamiliar public goods: preserving cold-water coral in Norway. Ecol. Econ. 112, 53–67.

Abelson, J., Forest, P.-G., Eyles, J., Smith, P., Martin, E., Gauvin, F.-P., 2003. Deliberations about deliberative methods: issues in the design and evaluation of public participation processes Social Science and Medicine. 57 pp. 239–251.

Adams, W.M., 2014. The value of valuing nature. Science 346, 549–551.

Aldred, J., 2006. Incommensurability and monetary valuation. Land Econ. 82, 141–161. Artioli, Y., Blackford, J.C., Nondal, G., Bellerby, R.G.J., Wakelin, S.L., Holt, J.T., Butenschon, M., Allen, J.I., 2014. Heterogeneity of impacts of high CO2 on the north western european shelf. Biogeosciences 11, 601–612.

Baker, R., Ruting, B., 2014. Environmental policy analysis: a guide to non-market valuation. Australian Government, Productivity Commission Staff Working Paper, Canberra.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R.E., Jenkins, P., Jefferiss, V., Jessamy, J., Madden, K., Munro, N., Myers, S., Naeem, J., Paavola, M., Rayment, S., Rosendo, J., Roughgarden, K.T., Turner, R.K., 2002. Economic reasons for conserving wild nature. Science 297, 950–953.

Blamey, R.K., Common, M.S., Quiggin, J.C., 1995. Respondents to contingent valuation surveys: consumers or citizens? Aust. J. Agric. Econ. 39, 263–288.

Böhnke-Henrichs, A., Baulcomb, C., Koss, R., Hussain, S.S., de Groot, R.S., 2013. Typology and indicators of ecosystem services for marine spatial planning and management. J. Environ. Manag. 130 135–14.

Börger, T., Beaumont, N.J., Pendleton, L., Boyle, K.J., Cooper, P., Fletcher, S., Haab, T., Hanemann, M., Hooper, T.L., Hussain, S.S., Portela, R., Stithou, M., Stockill, J., Taylor, T., Austen, M.C., 2014a. Incorporating ecosystem services in marine planning: the role of valuation. Mar. Policy 46, 161–170.

Börger, T., Hattam, C., Burdon, D., Atkins, J.P., Austen, M.C., 2014b. Valuing conservation benefits of an offshore marine protected area. Ecol. Econ. 108, 229–241.

Brannen, J., 2005. Mixing methods: the entry of qualitative and quantitative approaches into the research process. Int. J. Soc. Res. Methodol. 8 (3), 173–184.

Brown, T.C., 1984. The concept of value in resource allocation. Land Econ. 60, 231–246. Bryman, A., 2006. Integrating quantitative and qualitative research: how is it done? Qual. Res. 6, 97–113.

Burkhard, B., Kroll, F., Nedkov, S., Müller, F., 2012. Mapping ecosystem service supply, demand and budgets. Ecol. Indic. 21, 17–29.

Butenschön, M., Kay, S., 2013. Future scenarios of the biogeochemistry of the three regional seas FP7 – OCEAN- 2010. Project number 266445, VECTORS of Change in Oceans and Seas Marine Life, Impact on Economic Sectors. Deliverable D51.1 January 2013.

Carson, R.T., Hanemann, M.W., 2005. Contingent valuation. In: Mäler, K.-G., Vincent, J.R. (Eds.), handbook of environmental economics volume 2. Elsevier, pp. 821–936 Ch. 17.

Castro, A.J., Verburg, P.H., Martín-López, B., Garcia-Llorente, M., Cabello, J., Vaughn, C.C., López, E., 2014. Ecosystem service trade-offs from supply to social demand: a landscape-scale spatial analysis. Landsc. Urban Plan. 132, 102–110.

Christie, M., Fazey, I., Cooper, R., Hyde, T., Kenter, J.O., 2012. An evaluation of monetary and non-monetary techniques for assessing the importance of biodiversity and ecosystem services to people in countries with developing economies. Ecol. Econ. 83, 67–78.

Cook, G.S., Fletcher, P.J., Kelble, C.R., 2014. Towards marine ecosystem based management in south Florida: investigating the connections among ecosystem pressures, states, and services in a complex coastal system. Ecol. Indic. 44, 26–39.

Cooper, K., Burdon, D., Atkins, J.P., Weiss, L., Somerfield, P., Elliott, M., Turner, K., Ware, S., Vivian, C., 2013. Can the benefits of physical seabed restoration justify the costs? An assessment of a disused aggregate extraction site off the Thames estuary, UK. Mar. Pollut. Bull. 75, 33–45.

Corvalan, C., Hales, S., McMichael, A.J., Butler, C., Campell-Lendrum, D., Confaloneieri, U., Leitner, K., Lewis, N., Patz, J., Polson, K., Scheraga, J., Woodward, A., Younes, M., 2005. Ecosystems and human well-being: health synthesis: a report for the millennium ecosystem assessment. World Health Organisation, Geneva, Switzerland.

Crossman, N.D., Burkhard, B., Nedkov, S., Willemen, L., Petz, K., Palomo, I., Drakou, E.G., Martín-Lopez, B., McPhearson, T., Boyanova, K., Alkemade, R., Egoh, B., Dunbar, M.B., Maes, J., 2013. A blueprint for mapping and modelling ecosystem services. Ecosyst. Serv. 4, 4–14.

Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., Ricketts, T.H., Salzman, J., Shallenberger, R., 2009. Ecosystem services in decision making: time to deliver. Front. Ecol. 7 (1), 21–28.

Daily, G.C., Matson, P.A., 2008. Ecosystem services: from theory to implementation. Proc. Natl. Acad. Sci. U. S. A. 105, 9455–9456 PNAS.

Daily, G.C. (Ed.), 1997. Nature's services: societal dependence on natural ecosystems. Island Press, Washington DC.

Dasgupta, P., Levin, S., Lubchenco, J., 2000. Economic pathways to ecological sustainability. Bioscience 50, 339–345.

de Groot, R.S., Fisher, B., Christie, M., Aronson, J., Braat, L., Haines-Young, R., Gowdy, J., Maltby, E., Neuville, A., Polasky, S., Portela, R., Ring, I., 2010. Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation. In: Kumar, P. (Ed.), The economics of ecosystems and biodiversity (TEEB): ecological and economic foundations. Earthscan, London, pp. 9–40.

Defra, 2007. An introductory guide to valuing ecosystem services. Department for Environment, Food and Rural Affairs, London.

Defra, 2010. Incorporating valuation of ecosystem services into policy and project appraisal. Defra, London.

Diesing, M., Ware, S., Foster-Smith, R., Stewart, H., Long, D., Vanstaen, K., Forster, R., Morando, A., 2009. Understanding the marine environment – seabed habitat investigations of the Dogger Bank offshore draft SAC. JNCC Report No. 429. Joint Nature Conservation Committee, Peterborough 5 Appendices.

Douvere, F., Ehler, C.N., 2009. New perspectives on sea use management: initial findings from european experience with marine spatial planning. J. Environ. Manag. 90, 77–88.

Douvere, F., 2008. The importance of marine spatial planning in advancing ecosystem-based sea use management. Mar. Policy 32, 762–771.

EC, 1992. Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora (Habitats Directive). Off. J. Eur. Communities L 206, 7–59 22 July.

EU, 2014. Mapping and Assessment of Ecosystems and their Services Indicators for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020. Technical Report 2014 – 080.

Farber, S.C., Costanza, R., Wilson, M.A., 2002. Economic and ecological concepts for valuing ecosystem services. Ecol. Econ. 41, 375–392.

Fearon, J.D., 1998. Deliberation as discussion. In: Elster, J. (Ed.), Deliberative democracy. Cambridge University Press, Cambridge, pp. 44–68.

Fish, R., Burgess, J., Chilvers, J., Footitt, A., Haines-Young, R., Russel, D., Winter, D.M., 2011. Participatory and Deliberative Techniques to Embed an Ecosystems Approach into Decision Making: an Introductory Guide. Defra NR0124, London.

Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecol. Econ. 68 (3), 643–653.

Forewind, 2010. Dogger bank zonal characterisation report. Version 4 October 2010. Gómez-Baggethun, E., de Groot, R., Lomas, P.L., Montes, C., 2010. The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. Ecol. Econ. 69 (6), 1209–1218.

Greene, J.C., 2007. Mixed methods in social inquiry. Jossey-Bass, San Francisco, California. Greene, J.C., Caracelli, V.J., Graham, W.F., 1989. Towards a conceptual framework for mixed-method evaluation designs. Educ. Eval. Policy Anal.s 11, 255–274.

Groeneveld, R.A., Bosello, F., Butenschon, M., Elliot, M., Peck, M.A., Pinnegar, J.K., 2015. Defining scenarios of future vectors of change in marine life and associated economic sectors (submitted).

Hanley, N., Wright, R.E., Adamowicz, V., 1998. Using choice experiments to value the environment. Environ. Resour. Econ. 11, 413–428.

Hanley, N., Barbier, E.B., 2009. Pricing nature - cost-benefit analysis and environmental policy. Edward Elgar, Cheltenham.

Hansjürgens, B., Neßhöver, C., Schniewind, I., 2012. Der Nutzen von Ökonomie und Ökosystemleistungen für die Naturschutzpraxis. Workshop I: Einführung und Grundlagen. Bundesamt für Naturschutz, Bonn.

Hattam, C., Atkins, J.P., Beaumont, N., Börger, T., Böhnke-Henrichs, A., Burdon, D., de Groot, R., Hoefnagel, E., Nunes, P.A.L.D., Piwowarczyk, J., Sastre, S., Austen, M.C., 2015. Marine ecosystem services: linking indicators to their classification. Ecol. Indic. 49, 61–75.

Hattam, C., Börger, T., Garrard, S., Austen, M., Atkinson, J., Burdon, D., Piwowarczyk, J., Kedra, M., Weslawski, J.M., Delaney, A., Hadjimichael, M., Sastre, S., Canepa, A., Maynou, F., Piiol, L., Nunes, P.A.L.D., Loureiro, M., Voltaire, L., Serra, C., Otrachshenko, V., Bosello, F., Böhnke-Henrichs, A., de Groot, R., 2014. Impacts of change on ecosystem services and their values. FP7 – OCEAN- 2010. Project number 266445, VECTORS of Change in Oceans and Seas Marine Life, Impact on Economic Sectors. Deliverable D32.1 May 2014.

HM Treasury, 2003. The Green Book: Appraisal and Evaluation in Central Government. Treasury Guidance. London TSO. https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/220541/green_book_complete.pdf accessed 06/08/15.

Howarth, R.B., Wilson, M.A., 2006. A theoretical approach to deliberative valuation: aggregation by mutual consent. Land Econ. 82, 1–16.

Hufnagl, M., Peck, M.A., Nash, R.D.M., Pohlmann, T., Rijnsdorp, A.D., 2013. Changes in potential north sea spawning grounds of plaice (pleuronectes platessa L.) based on early life stage connectivity to nursery habitats. J. Sea Res. 84, 26–39.

Huitema, D., van de Kerkhof, M., Pesch, U., 2007. The nature of the beast: are citizens' juries deliberative or pluralist? Policy. Sci. 40, 287–311.

Hynes, S., Tinch, D., Hanley, N., 2013. Valuing improvements to coastal waters using choice experiments: an application to revisions of the EU bathing waters directive. Mar. Policy 40, 137–144.

IPCC, 2000. Special Report on Emissions Scenarios (SRES). In: Nakicenovic, N., Swart, R. (Eds.), International Panel on Climate Change. Cambridge University Press, Cambridge, UK, p. 612.

IPIECA, 2011. Ecosystem services guidance — biodiversity and ecosystem services guide and checklists. OGP Report No. 461, p. 31.

JNCC, 2011. Dogger Bank SAC Final Impact Assessment. 4. Joint nature conservation committee, Peterborough, UK July 2011.

Jobstvogt, N., Hanley, N., Hynes, S., Kenter, J., Witte, U., 2014. Twenty thousand sterling under the sea: estimating the value of protecting deep-sea biodiversity. Ecol. Econ. 97, 10–19.

Kallis, G., Gómez-Baggethun, E., Zografos, C., 2013. To value or not to value? That is not the question. Ecol. Econ. 94, 97–105.

Kenter, J.O., Bryce, R., Davies, A., Jobstvogt, N., Watson, V., Ranger, S., Solandt, J.L., Duncan, C., Christie, M., Crump, H., Irvine, K.N., Pinard, M., Reed, M.S., 2013. The value of potential marine protected areas in the UK to divers and sea anglers. UNEP-WCMC, Cambridge, UK.

Kosoy, N., Corbera, E., 2010. Payments for ecosystem services as commodity fetishism. Ecol. Econ. 69 (6), 1228–1236.

Krinsky, I., Robb, A.L., 1986. On approximating the statistical properties of elasticities. Rev. Econ. Stat. 68, 715–719.

Laurans, Y., Rankovic, A., Billé, R., Pirard, R., Mermet, L., 2013. Use of ecosystem services economic valuation for decision making: questioning a literature blindspot. J. Environ. Manag. 119, 208–219.

Lester, S.E., McLeod, K.L., Tallis, H., Ruckelshaus, M., Halpern, B.S., Levin, P.S., Chavez, F.P., Pomeroy, C., McCay, B.J., Costello, C., Gaines, S.D., Mace, A.J., Barth, J.A., Fluharty, D.L., Parrish, J.K., 2010. Science in support of ecosystem-based management for the US West Coast and beyond. Biol. Conserv. 143, 576–587.

Liquete, C., Piroddi, C., Drakou, E.G., Gurney, L., Katsanevakis, S., Charef, A., Egoh, B., 2013. Current status and future prospects for the assessment of marine and coastal ecosystem services: as systematic review. PLoS ONE 8, e67737.

Loomis, J., Santiago, L., 2013. Economic valuation of beach quality improvements: comparing incremental attribute values estimated from two stated preference valuation methods. Coast. Manag. 41, 75–86.

Lopes, R., Videira, N., 2013. Valuing marine and coastal ecosystem services: an integrated participatory framework. Ocean Coast. Manag. 84, 153–162.

Louviere, J.J., Hensher, D.A., Swait, J.D., 2000. Stated choice models. Analysis and application. Cambridge University Press, Cambridge.

Liu, S., Costanza, R., Farber, S., Troy, A., 2010. Valuing ecosystem services. Ann. N. Y. Acad. Sci. 1185, 54–78.

Mackinson, S., Curtis, H., Brown, R., McTaggart, K., Taylor, N., Neville, S., et al., 2006. A report on the perceptions of the fishing industry into the potential socio-economic impacts of offshore wind energy developments on their work patterns and income. Science series technical report 133. CEFAS, Lowestoft, p. 60.

Martinez-Alier, J., Munda, G., O'Neill, J., 1998. Weak comparability of values as a foundation for ecological economics. Ecol. Econ. 26 (3), 277–286.

Martín-López, B., Gómez-Baggethun, E., García-Llorente, M., 2014. Trade-offs across value-domains in ecosystem service assessment. Ecol. Indic. 37, 220–228.

Martín-López, B., García-Llorente, M., Gómez-Baggethun, E., Montes, C., 2010. Evaluación de los servicios de los ecosistemas del sistema socio-ecológico de doiana. Foro de Sostenibilidad 4, 91–111.

McVittie, A., Moran, D., 2010. Valuing the non-use benefits of marine conservation zones: an application to the UK marine bill. Ecol. Econ. 70, 413–424.

MMO, 2014. Strategic evidence plan. Marine Management Organisation, Newcastle available at: https://www.gov.uk/government/publications/strategic-evidence-plan, last accessed December 2014.

Mooney, H., Cooper, A., Reid, W., 2005. Confronting the human dilemma: how can ecosystems provide sustainable services to benefit society? Nature 434, 561–562.

NSRAC, 2012. Position paper on fisheries management in relation to nature conservation, including a zoning proposal, for the combined area covered by the 3 national Natura 2000 sites (SACs) on the Dogger Bank http://nsrac.org/wp-content/uploads/2012/07/ NSRAC-1112-7-2012-04-09-Dogger-Bank-SACs-Position-Paper-FINAL.pdf.

Orr, S.W., 2007. Values, preferences, and the citizen-consumer distinction in cost-benefit analysis. Politics Philos. Econ. 6, 107–130.

Parks, S., Gowdy, J., 2013. What have economists learnt about valuing nature? A review essay. Ecosyst. Serv. 3, e1–e10.

Pascual, M., Borja, A., Franco, J., Burdon, D., Atkins, J.P., Elliott, M., 2012. What are the costs and benefits of biodiversity recovery in a highly polluted estuary? Water Res. 46, 205–217.

Pascual, M., Borja, A., Vanden Eede, S., Deneudt, K., Vincx, M., Galparsoro, I., Legorburu, I., 2011. Marine biological valuation mapping of the basque continental shelf (bay of biscay), within the context of marine spatial planning. Estuar. Coast. Shelf Sci. 95 (1), 186–198.

Pearce, D., Atkinson, G., Mourato, S., 2006. Cost-benefit analysis and the environment. Recent developments, OECD, Paris.

Phal-Wostl, C., 2007. The implications of complexity for integrated resources management. Environ. Model. Softw. 22, 561–569.

Pike, K., Wright, P., Wink, B., Fletcher, S., 2014. The assessment of cultural ecosystem services in the marine environment using Q methodology. J. Coast. Conserv. 1–9 http://dx.doi.org/10.1007/s11852-014-0350-z.

Ressurreição, A., Gibbons, J., Bentley, C., Burdon, D., Atkins, J.P., Kaiser, M., Austen, M., Santos, R., Dentinho, T.P., Zarzycki, T., Edwards-Jones, G., 2012. Different cultures, different values: the role of cultural variation in public's willingness to pay for marine species conservation. Biol. Conserv. 145, 148–159.

Richardson L., Loomis J., Kroeger T. and Casey F. 2014. The role of benefit transfer in ecosystem service valuation. Ecol. Econ., DOI: http://dx.doi.org/http://dx.doi.org/10.1016/j.ecolecon.2014.02.018.

Szabó, Z., 2011. Reducing protest responses by deliberative monetary valuation: improving the validity of biodiversity valuation. Ecol. Econ. 72, 37–44.

Spash, C.L., 2007. Deliberative monetary valuation (DMV): issues in combining economic and political processes to value environmental change. Ecol. Econ. 63, 690–699.

Stagl, S. 2004. Valuation for sustainable development – the role of multicriteria evaluation. Vierteljahrshefte zur Wirtschaftsforschung 73 (1): S. 53–62138

Tallis, H., Kareiva, P., Marvier, M., Chang, A., 2008. An ecosystem services framework to support both practical conservation and economic development. Proc. Nat. Acad. Sci. U S A 105, 9457–9464.

Teddlie, C., Tashakkori, A., 2009. Foundations of mixed methods research: integrating quantitative and qualitative approaches in the social and behavioural sciences. Sage Publications Inc., California.

Townsend, M., Thrush, S.F., Lohrer, A.M., Hewitt, J.E., Lundquist, C.J., Carbines, M., Felsing, M., 2014. Overcoming the challenges of data scarcity in mapping marine ecosystem service potential. Ecosyst. Serv. 8, 44–55.

Travers, A., 1990. The invisible woman: a feminist critique of Habermas' theory of communicative action University of British Columbia. MSc Thesis. Accessed online: https://circle.ubc.ca/handle/2429/29857.

Wattage, P., Glenn, H., Mardle, S., Van Rensburg, T., Grehan, A., Foley, N., 2011. Economic value of conserving deep-sea corals in Irish waters: a choice experiment study on marine protected areas. Fish. Res. 107, 59–67.

Wilson, M.A., Howarth, R.B., 2002. Discourse-based valuation of ecosystem services: establishing fair outcomes through group deliberation. Ecol. Econ. 41, 431-443.