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Prospects for Valuation in Marine Decision Making in Europe

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Prospects for Valuation in Marine Decision Making in Europe

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1. INTRODUCTION

Globally, marine systems face what Waltham *et al.* (2020) call "a triple whammy" of increasing industrialization and urbanization, increasing loss of biological and physical resources, and decreasing resilience due to the consequences of climate change. Even Europe, with one of the most comprehensive frameworks for marine environmental protection, has reported that while some marine species show signs of recovery, others show steep deterioration with further measures needed to tackle ongoing pressures including overfishing, physical disturbance, plastic litter, pollution/eutrophication and underwater noise (EC, 2020a). This is a major policy issue.

The UN Decade on Ecosystem Restoration (2021-2030) coincides with the Decade of Ocean Science for Sustainable Development, the deadline for the achievement of the Sustainable Development Goals (SDGs) and the timeline scientists have identified as the last chance to prevent catastrophic climate change.[1] This marks high-level recognition that the SDGs can only be achieved if the decline of ecosystems and biodiversity can be halted and reversed. This will require effective control of pressures (both ongoing, *e.g.*, fishing, and novel, *e.g.*, deep sea mining), meaningful protection and enforcement of protected areas, and significant investments in ecosystem restoration. This paper explores the important role of economic valuation and appraisal in achieving these goals in marine systems, through identifying and quantifying the economic and social consequences of marine exploitation and restoration activities. Recognising the services provided by healthy ecosystems, quantifying them, and valuing the benefits to society from those services helps decision makers to take such values into account when assessing policies and priorities.

In what follows the European policy framework driving the marine conservation and restoration agenda is presented. Section 3 then reviews the tools of economic valuation and appraisal while section 4 considers the use to date of cost-benefit analysis (CBA) for marine ecosystem management in decision contexts associated with marine protection and restoration. The final section presents a critique of the use of economic valuation and appraisal for marine environment decision making and draws conclusions for European policy.

[1] https://www.decadeonrestoration.org/about-un-decade

2. EUROPEAN POLICY: FAILURE AND OPPORTUNITIES

The EU Biodiversity Strategy to 2030 (EUBS2030) follows a failure to meet the 2020 targets, noting that "significant implementation and regulatory gaps hinder progress." Although the Natura 2000 network (core breeding and resting sites for rare and threatened species in the EU that is planned for expansion as part of EUBS2030) helps maintain ecosystem condition and biodiversity in surrounding areas "pressures remain high and the conservation measures undertaken are still insufficient" (EEA, 2019). Indeed, it has long been clear that biodiversity and restoration targets cannot be achieved solely through the protected area network, important though that is. The European Habitat Forum assessment notes on the positive side that the knowledge base has increased, but that ecosystem degradation continues, with a lack of strategic planning, unclear baselines, no commitment to specific restoration targets, and insufficient investment in restoration and green infrastructure (European Habitat Forum, 2020).

Specifically in the marine environment, the EU Marine Strategy Framework Directive (MSFD) aims to achieve Good Environmental Status (GES) of the EU's marine waters. This target was supposed to be met by 2020. However, despite much success, the implementation report (EC, 2020a) notes that "the biodiversity of marine ecosystems is still vulnerable in Europe's seas and the good state of habitats and species is not secured." A review is due by 2023 and should provide more detail on successes and failures, and the ongoing relevance of the MSFD in the context of the EUBS2030, the EU Zero Pollution Action Plan and the EU Climate Adaptation Strategy.[2]

The higher-level reasons behind the failures to halt biodiversity loss and achieve GES across the marine environment lie in well-known problems, including weak implementation of existing legislation, failure to mainstream biodiversity and environmental concerns across all policy sectors, and insufficient resources for conservation coupled with the failure to reform perverse subsidies (Zito *et al,* 2019). Behind them lie a lack of political will to take nature loss seriously and opposition from stakeholders with vested interests in the status quo. This reflects global and European pressures that have continued to marginalise environmental objectives in EU policy making, including the global and euro financial crises, failure to meet the Lisbon goals, migration and energy security concerns, rising populism, differentiated integration between member states (including Brexit) and the

[2] https://ec.europa.eu/environment/news/commission-publishes-msfd-roadmap-2021-04-09_en

disconnect between the longer-term environmental challenges and the short-term exigencies of electoral politics. These problems persist, and the economic and social impacts of the current pandemic are likely to exacerbate them. And these are the problems that the EUBS2030 must solve if it is to meet the pressing need, identified in the latest "State of Nature in the EU" report "*for a step-change in action if we are to have any serious chance of putting Europe's biodiversity on a path to recovery by 2030*" (EC, 2020b).

The EUBS2030 sits within the "European Green Deal", Europe's new agenda for sustainable economic growth across the region. Other components of the Green Deal include, inter alia, the Green Deal Investment Plan,[3] the Just Transition Mechanism,[4] a proposed European Climate Law, and a new Circular Economy Action Plan. The Green Deal frames the problems of climate change and environmental degradation as "an existential threat to Europe and the world", for which "Europe needs a new growth strategy that will transform the Union into a modern, resource-efficient and competitive economy"[5].

The tension between environmental and growth objectives here is clear, as is the disconnect between "climate" and "other environmental issues". The main focus of the Green Deal is climate neutrality by 2050, with decoupling of growth from resource use, and social justice ("no person and no place is left behind"); ecosystems are seen as solutions "not only to protect biodiversity but also to enhance carbon uptake and contribute to climate change mitigation as well as to deliver essential benefits to people, agriculture, and the economy." (Maes 2020).

The recent EU Parliament resolution on the EU Forest Strategy[6] illustrates how these objectives are being translated to policy. It invites the Commission to "explore options to incentivise and remunerate climate, biodiversity and other ecosystem services appropriately", and stresses "the importance of developing and ensuring a market-based bio-economy in the EU". The EUBS2030 will also have to work within that framework: ensuring sustainability will require strong evidence, and

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[3] The European Green Deal Investment Plan (EGDIP), also referred to as Sustainable Europe Investment Plan (SEIP), is the investment pillar of the Green Deal. To achieve the goals set by the European Green Deal, the Plan will mobilise at least €1 trillion in sustainable investments over the next decade.
[4] The Just Transition Mechanism (JTM) is a key tool to ensure that the transition towards a climate-neutral economy happens in a fair way, leaving no one behind. It provides targeted support to help mobilise at least €150 billion over the period 2021-2027 in the most affected regions, to alleviate the socio-economic impact of the transition.
[5] https://ec.europa.eu/info/strategy/priorities-2019-2024/european-green-deal_en.
[6] https://www.europarl.europa.eu/doceo/document/TA-9-2020-0257_EN.html.

preferably monetary valuation, to recognise and internalise the importance of ecosystem services that at present are not reflected in markets.

3. ECONOMIC TOOLS FOR SUPPORTING POLICY

The internalisation of the importance of non-market ecosystem services started with ecosystem services classification frameworks, developed since the late 1990s (Daily 1997) through the Millennium Ecosystem Assessment (see Reid *et al.*, 2005) and subsequent contributions (*e.g.*, Silvestri and Kershaw, 2010; Turner and Daily, 2008; Boyd and Banzhaf, 2007; TEEB, 2010). More recently, the European Environment Agency (EEA) has led work to develop the Common International Classification of Ecosystem Services (CICES) and the US Environment Protection Agency (EPA) has developed the Final Ecosystem Goods and Services Classification System (FEGS-CS). The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) was established in 2012 to assess the state of biodiversity and ecosystem services, in response to requests from decision-makers. Large amounts of data are now available for many services: Caro *et al.* (2018) review 581 open access databases related to indicators of coastal and marine ecosystem services.

While there remain differences of interpretation and application, notably regarding the appropriate treatment of supporting or intermediate services and over the use of alternative framings such as ‘nature’s contributions to people’ (Pascual *et al.*, 2017), it is fair to say that the basic idea of classifying the ways in which human wellbeing depends directly and indirectly on natural environments is now mainstream and widespread. However, this does not carry directly to the ways in which these services are assessed, valued and incorporated in appraisal and decision-making processes.

3.1 ECONOMIC APPRAISAL

There is an unequivocal obligation under the United Nations Convention for the Law of the Sea (UNCLOS) to protect and preserve the marine environment, as well as obligations of a more procedural nature such as using best environmental practices and ensuring transparency and participation (Billett *et al.*, 2015). Article 2008 requires that national laws and regulations be “no less effective than” the international rules. Elsewhere, Annex III of the EU MSFD was amended in 2017 to better link ecosystem components, anthropogenic pressures and impacts on the marine environment with the MSFD's 11 descriptors and with the new Decision on Good Environmental Status (EC, 2017). But despite the obligation to protect and preserve, and to reach GES, there remain difficult choices regarding how exactly that is to be achieved, including issues regarding assessment of cost-effectiveness of alternative measures to achieve or maintain GES of Member State marine waters and the demonstration of 'disproportionate costs'.[7]

There is a long history of using economic tools for decision support at the economy-environment interface (Watkiss *et al.*, 2014) including financial appraisal, economic impact assessment (EcIA), multi-criteria assessment (MCA) cost-effectiveness analysis (CEA) and cost-benefit analysis (CBA), as well as strategic approaches to risk management at a strategic level via Portfolio Analysis, Real Option Analysis (ROA) and Iterative Risk Management. These tools are not mutually exclusive and often will be used together, and alongside environmental impact assessment (EIA) which is required for a wide range of marine decisions. Financial appraisal is also needed for most proposed investments, to assess cashflow and overall profitability for potential investors, operators, and public bodies. EcIA goes further by estimating direct, indirect and induced changes of output, gross value added, employment and tax revenues resulting from a project or industry. It is commonly used in regional and national policy appraisal procedures.

However, these methods focus only on the market impacts. Methods such as MCA, CEA and CBA take a broader view, aiming to assess other impacts on the environment and human welfare. In the case of CBA and CEA, this involves monetary valuation of non-marketed goods and services, using the TEV (Total Economic Value) framework of welfare values that includes use value, non-use value, option value and bequest value (Plottu and Plottu 2007). CBA compares all the benefits and costs of project/policy options that can be valued in monetary terms, weighted by their probabilities, discounted to convert future values to present-day equivalents, then aggregated to give expected net present value (Boardman *et al.*, 2017; OECD, 2018). The method depends on being able to quantify all the impacts of project options (states of the world with and without the option) and on being able to ascribe robust monetary values to each impact (Watkiss *et al.*, 2014). CBA can compare options for a specific decision, and rank/prioritise them in terms of their net present value (NPV), benefit: cost ratio (BCR), or internal rate of return (IRR). In principle all projects with positive net present value are

[7] "The most challenging areas were pointed out by respondents to be Cost-Effectiveness Analysis (CEA) of measures and justification of disproportionate costs" – lessons learned from the WFD, from Working Group on Economic and Social Assessment Economic and Social Analysis for the Initial Assessment for the Marine Strategy Framework Directive: A Guidance Document. https://circabc.europa.eu/sd/a/bdcafa98-1ede-4306-997e-ec2d991dcb6f/2.3b-%20ESA%20Guidance.doc

socially efficient, however where resources are limited CBA results can help to rank interventions in order of priority – as well as make an argument for increased funding overall. Extensive official guidance for appraisal exists in many jurisdictions (*e.g.*, EU (EC, 2015), UK (HMT, 2020), US (US EPA, 2010)), and specific guidance for value transfer is also available (*e.g.*, eftec, 2010).

In practice, CBA rarely (if ever) covers all impacts in monetary terms, with non-monetised items often being reported separately. CBA almost always involves use of value transfer from existing valuation studies, although that is considered contentious by some (*e.g.*, Ravenscroft, 2019)[8]. Furthermore, monetary valuation does not capture everything of importance to society: economic impacts, distributional effects, and environmental justice should also be considered (Fonner *et al.*, 2020). Risk is generally treated by summing expected values using a probability distribution of outcomes (risk-averse aggregating functions are also possible). But the expected value approach may give inadequate weight to high-consequence, low-probability outcomes (Taleb *et al.*, 2014) and in practice, full probability distributions are often lacking.

For these and other reasons, CBA should be seen as a tool for structuring information and for supporting decisions, not as a substitute for deliberation or a decision-making tool. Other methods can be used alongside CBA. Qualitative analysis may be used to highlight different sources of uncertainty and develop scenarios; quantitative analysis can then be applied to each scenario and results compared (see *e.g.* Eory *et al.*, 2014). Another option is to move focus away from maximising expected NPV, for example by seeking to minimise the expected cost of error, which requires subjective probabilities for each scenario (Hallegatte *et al.*, 2011). Non-probabilistic methods can focus on other approaches to robust policy development, while still drawing on estimates of costs and benefits under different scenarios.

CEA is an alternative where a target can be defined but not valued. CEA then seeks to establish the most cost-effective means of achieving the goal. The advantage is that it is not necessary to value the benefits, which is often the most challenging or costly aspect of CBA. However, ancillary benefits that vary across options should in principle be included. CEA is widespread in climate change

[8] A recent study by Hynes *et al.* (2021b) did find however that preferences and willingness to pay estimates from a repeated stated preference discrete choice experiment concerning marine ecosystem services would appear to remain relatively stable even through a global pandemic suggesting temporal reliability of welfare estimates for use in value transfer.

economics, because it is very difficult to estimate the benefits of mitigation, but easy to derive a common metric of it. However, it is harder to apply in areas such as climate adaptation or ecosystem protection or restoration, because there is no common metric for ‘how much’ adaptation, protection or restoration has been achieved. One partial solution to this problem is use of ecological equivalence assessment methods (Bezombes *et al.*, 2017; Bas *et al.*, 2016) although these can contain important and contentious economic and ecological assumptions (Desvousges *et al.*, 2018). Equivalency analysis has been used in the EU for biodiversity offsets under the Habitat Directive, where compensation was required to mitigate for projects that damaged or destroyed protected habitats (Quétier and Lavorel, 2011). The Environmental Liability Directive (ELD) established a need for more rigorous quantification in the application of equivalency methods (Desvousges *et al.*, 2018), with environmental damage being defined with respect to the contentious legal concept of favourable conservation status (FCS) (Epstein *et al.*, 2015). An alternative to using arbitrary weights applied to a suite of indicators would be to focus on the ecosystem services provided by the system, and aggregate these using monetary values, giving weights that reflect the relative importance of different services, including in spatially explicit ways, and directly comparable to costs.

3.2 MONETARY VALUATION OF ECOSYSTEM SERVICES

Various tools of environmental valuation are widely used to incorporate environmental impacts in appraisal methods, and for the purposes of environmental and ecosystem accounting. There is a large and growing literature of original economic valuation studies, meta-analyses of economic valuation studies (*e.g.*, Brouwer *et al.*, 1999; Brander *et al.*, 2012) and economic valuation databases, notably the TEEB (The Economics of Ecosystems and Biodiversity) database (Van der Ploeg and de Groot, 2010; Van der Ploeg *et al.*, 2010) and its applications to global valuation (de Groot *et al.*, 2012; Costanza *et al.*, 2014). Most recently, international standards ISO14007 (Environmental management — Guidelines for determining environmental costs and benefits) and 14008 (Monetary valuation of environmental impacts and related environmental aspects) have been published. There are also International Statistical Standards for environmental accounting (the System of Environmental-Economic Accounting Central Framework, SEEA-CF) and ecosystem accounting (SEEA Ecosystem Accounts, SEEA-EEA), the latter having been refined via the Virtual Expert Forum on SEEA Experimental Ecosystem Accounting 2020 and associated research prior to formal adoption by the UN Statistical Commission in March 2021 (Obst *et al.*, 2020).

The knowledge base for mapping and valuing ecosystem services in Europe has been further developed through the EUBS2020 which called (Action 5) for Member States to "map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020" (EC, 2011). This led to a sustained research effort in ecosystem service assessment, valuation, and reporting, through MAES (Mapping and Assessment of Ecosystems and their Services) (Maes *et al.*, 2020) and KIP-INCA (Knowledge Innovation Project - Integrated system for Natural Capital and ecosystem services Accounting)[9] and supporting research projects. KIP-INCA in particular aims to develop natural capital accounts to understand dependence on ecosystems at multiple levels: macro-indicators to use alongside GDP, support for EU Sectoral policies, promoting environmentally responsible business practices, and contributing to the SEEA-CF and -EEA accounting standards (EC, 2019). Although the first EU-wide ecosystem assessment (Maes *et al*., 2020) did not include any monetary estimates, it lays the foundations for ecosystem service quantification and valuation at the European scale.

At the same time, KIP-INCA has made progress on the use of monetary values within an accounting framework. For example, the experimental seagrass accounts pilot study[10] reviews 12 studies that focus on four services provided by seagrass: carbon sequestration/storage; shoreline protection; fish nursery and habitat; and nutrient cycling. Valuations are mostly based on market prices or costs, "commensurate with the exchange value approach required for integration with national accounts." Exchange values are measured as the product of market prices and quantities to give the total value of income, production and expenditure in transactions. The best way in which to derive proxy exchange values for non-market goods and services for use in accounting remains a live research issue (Atkinson and Obst, 2017; Caparrós *et al.*, 2017; Grimsrud *et al.*, 2018), with a focus on ensuring comparability within an exchange value framework.

Welfare values differ from exchange values by including consumer surplus but excluding production costs. Welfare value therefore gives a theoretically valid measure of economic value, while exchange value does not (Brouwer *et al.*, 2013). Nevertheless, exchange values are used in SEEA-EEA to ensure compatibility within the System of National Accounts (SNA) framework, the underpinning for

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[9] https://ec.europa.eu/environment/nature/capital_accounting/pdf/MAES_INCA_2019_report_FINAL-fpub.pdf
[10]https://ec.europa.eu/environment/nature/capital_accounting/pdf/Seagrass%20Marine%20Accounts.pdf

GDP estimates, which measures transactions (incomes and expenditures). Although GDP is often thought of as a measure of welfare, and growth in GDP is a strong political and policy priority, there are several conventions in the SNA that argue against the welfare interpretation of the accounts (SNA, 2008). The values used for accounting purposes are not the same ones that should be used in CBA or other appraisal focused on changes in human welfare: otherwise, exclusion of non-use values and consumer surplus would tend to lead CBA to favour industrial development over conservation, and favour traded services over public goods.

3.3 STATED PREFERENCE VALUATION

How welfare values can be estimated depends on the good or service of interest. Market evidence is generally preferred if available (*e.g.*, demand for fish), while for non-market goods revealed preference methods using evidence on actual behaviour is sometimes possible (*e.g.*, travel cost modelling of recreation values). Proxy methods such as avoided or replacement costs (*e.g.*, flood damages prevented, or costs of providing man-made flood protection) are also common, though not strictly correct as measures of welfare. In many cases the only viable option is stated preference surveys. The applicability of stated preference methods to marine environments can be hampered by the unfamiliarity of most people with marine ecosystems and their components, especially for the deep sea (Hanley *et al.*, 2015). This can spill over to lower confidence in these values from decision makers. Few CBAs include values for the non-use aspects, and those that do generally treat the uncertainty, and risk of double counting, with considerable care and conservatism, even to the point of separating them from the overall assessment: McVittie and Moran (2010) present them as an alternative way of reaching the conclusion that protection is justified, either by the direct ecosystem service values alone, or by non-use alone. Strategically, this works, but in more marginal cases a positive net present value may depend on summing the use and non-use values. And stated preference methods are effectively the only approaches that may be able to shed light on potentially high public non-use values for marine conservation. A few studies have attempted to apply stated preference methods to protection of deep-sea ecosystems (Chen *et al.*, 2021; Hynes *et al.*, 2021a; O'Connor *et al.*, 2020; Jobstvogt *et al.*, 2014; Wattage *et al.*, 2011; Aanesen *et al.*, 2015) with results that suggest non-use values could be significant.

However, there are many well-recognised issues with regards to stated preference methods, as well as guidance on how to minimise the problems through conservative study design to achieve reliable results (Arrow *et al.*, 1993; Bateman *et al.*, 2002; Johnston *et al.*, 2017). Three issues are particularly important for applications to marine conservation. Firstly, it is not always clear exactly what impacts are covered by the responses: there is a risk that respondents consider not only the specified change, but also changes that may be perceived as linked (for example general improvements in biodiversity conservation or general environmental quality, or changes at much greater spatial scale than the specific project) leading to a risk of double-counting.

Secondly, the closely related issue of scope insensitivity can be particularly tricky for applications to restoration, where the areas to be treated can seem large in absolute terms while at the same time being tiny in proportion to the marine ecosystems in which they are located. Furthermore, although recovering and restored ecosystems have less abundance, diversity and function than ‘undisturbed’ ecosystems (Benayas *et al.*, 2009), and accumulate an interim recovery debt even if complete restoration is feasible (Moreno-Mateos *et al.*, 2017), studies find higher WTP for removal and restoration operations than for preservation and prevention of further biodiversity loss (Tonin, 2019). This could be because respondents perceive prevention and control activities as being embedded in restoration, or because respondents place a premium on active over passive approaches. This is likely if marine environments have ‘protected value’ or ‘sacred value’ status (Gibson *et al.*, 2016) triggering deontological reactions and a preference for action over passivity (Tanner *et al.*, 2008). One implication of deontological thinking is that people become less sensitive to the magnitude of outcomes: they focus on the inherent wrongness or rightness of the activity (Gibson *et al.*, 2016). In other words, applications to conservation problems involving ‘protected values’ is precisely the situation in which we would expect scope insensitivity to arise. People may also be too focused on short-term improvements: for example, Lewis *et al.* (2019) found that the public placed significant value on achieving Pacific salmon recovery more quickly.

Finally, there can be a problem associated with timing of benefits and stated preference responses. Generally, in SP research it is accepted that there is a problem with ‘recontracting’ if asking people to give WTP amounts for a long run of years (for example a tax increase every year for 10 years) so it is often considered preferable to focus on one-time payments. However, conservation activities may require many years to take effect, and inevitably studies often relate to long-term protection of species or resources. There can be ambiguity in determining exactly what period of payment and benefit SP responses refer to, and there is a risk of some studies contrasting (in the most extreme case) annual flows of conservation costs with capitalised present values of benefits.

The above risks can be minimised by careful survey design and debriefing, and full reporting on the associated assumptions, tests for scope sensitivity and so on.

Further research in this area would be useful in determining the best ways of conducting and interpreting stated preference surveys for assessing the non-use benefits of marine conservation.

3.4 DEALING WITH UNCERTAINTY

In standard economic appraisal, risk and time are addressed using expected values and discounting. However, in the context of marine exploitation/restoration both are particularly important, due to very long timescales, serious information constraints, and the potential for catastrophic or irreversible outcomes. Specifically for restoration, uncertainty about the chances of success can "profoundly affect expected benefits" (Wainger *et al.*, 2017).

Calculating expected values requires knowledge of the probability distributions of different outcomes. Where multiple sources of risk exist, Monte Carlo methods are often used to build up distributions of overall outcomes. However, in many situations we are facing not (measurable) risk but (immeasurable) uncertainty, or ignorance (where not all possible outcomes are known, following Knight 1921:19), so expected values cannot be calculated. In addition, since much of the uncertainty relates to lack of knowledge regarding marine processes and ecology, the probabilities associated with specific impacts and sites are not independent, and so would not tend to 'average out' across assessments. Krutilla *et al.* (2021) discuss recent focus on "deep" or "fundamental" uncertainties, that mean it may be impossible to agree on crucial components of decision-making, such as appropriate models of system interactions, the full range of outcomes, or how to evaluate them (Lempert *et al.*, 2003; Walker *et al.*, 2013). Krutilla *et al.* (2021) explain that a "predict-then-act" decision structure is suited to situations of "risk" but seriously limited where deep uncertainty or ambiguity prevent predictive modelling and where consequences of error may be substantial and/or irreversible. Taleb *et al.* (2014) discuss the "fat tails" arising under high model uncertainty and explain that this should motivate more precautionary policies, because an increased chance of ruinous outcomes is much more policy-relevant than the increased chance of happy surprises (a fact not reflected in the 'expected' outcome).

Ounanian *et al.* (2018) draw a slightly different distinction among incomplete knowledge, unpredictability, and ambiguity, the latter defined as "uncertainty due to the presence of multiple knowledge frames or different but (equally) sensible interpretations of the same phenomenon, problem or situation". If decision-maker reluctance to "believe" valuation studies, or to accept that people have non-use values, is related to ambiguity in this sense, it will not be resolved simply by doing more/better valuation studies.

CBA does not have to be limited to expected values, and can (should) include ranges, scenarios and sensitivity analysis to important risks and uncertainties. Well-presented CBA information will demonstrate the full range of possible value outcomes, including possible thresholds and very damaging outcomes, not just a 'best estimate'. And CBA should only ever be considered as one part of a decision process – as a way of structuring and presenting information, but not a replacement for deliberation. Furthermore, supporting decisions about natural resource management in the "predict then act" context is only one reason for CBA: it can also be used ex-post for evaluating actual outcomes, for example, or for informing compensation levels. Where the CBA is not intended to feed into decisions with direct impacts on resource management and conservation, the consequences of "getting it wrong" will be less severe. Nevertheless, these concerns about deep uncertainty are well-founded and critical to the appropriate use of CBA in different contexts.

3.5 TIME AND DISCOUNTING

The use of discounting in appraisal makes costs and benefits far in the future much less important than present costs and benefits. There is substantial debate concerning the appropriate use of discounting for ecosystem services, in particular for the far future. A US EPA expert panel of 12 economists (Arrow *et al.*, 2013) unanimously agreed that "the Ramsey formula provides a useful framework for thinking about intergenerational discounting" but did not reach agreement on how the parameters of the Ramsey formula might be determined empirically, let alone on actual values. Discount rates of a few percent, standard for short-term policy appraisal, result in huge discounting of long-term impacts. Some authors advocate declining or hyperbolic discount rates (Kirby, 1997; HMT, 2020) to combat this problem. Others use a low constant rate (*e.g.*, Stern, 2006), and Moxnes (2014) reports evidence that, when very long-term sustainability of well-being is threatened, most people's implicit discount rates do resemble these low estimates.

Heal and Millner (2014) argue that there are no objectively correct discount rates, just different ethical positions that should all be considered, aggregating "the diverse preferences of individuals into a representative discount rate". But there is no way to know the preferences of most of the individuals involved, namely future generations. Weitzman (2007) instead shifts the focus from "consumption smoothing" to one of "how much insurance to buy to offset the small chance of a ruinous catastrophe that is difficult to compensate by ordinary savings." Moxnes (2014) poses the question "could one do without welfare functions and discounting when choosing between policies?", and reports that people presented with graphs of policy consequences over time are indeed able to make consistent choices. Some positive discount rate is essential to CBA, but again, this should be seen in the context of one input to decision processes.

3.6 SCALE AND VALUE

Scale is another important factor that should be reflected in the non-linearity of values. There are many services for which values are not a simple linear function of the area of an ecosystem, including coastal flood protection and recreation values (Barbier 2007; Koch *et al.*, 2009). More generally, the relationship between ecosystem services (quantity and/or quality) and their value will depend on their scarcity with respect to demand (Farley, 2008). Any value estimate is only valid at the margin, and any value for a non-marginal change in quantities is usually going to be an integral of a non-constant function. The severity of error associated with imprecise valuation depends on the rate at which that function changes (*i.e.*, the elasticity of demand): risks are low where elasticity is low; where elasticity is high, rapidly changing values make the consequences of small quantity changes significant, so valuation and market-based instruments are riskier; for 'critical natural capital', elasticity is effectively infinite, marginal valuation is inappropriate, and the Precautionary Principle must apply.

At oceanic scales, we might expect quite strong non-linearity, whereas at the scale of individual exploitation/restoration sites, any thresholds faced are more likely to be local. However, the high level of connectivity in marine environments weakens this proposition: for example, specific areas could be crucial links in ocean chains of larval dispersal and settlement. There is also a potential problem associated with independent valuation of lots of separate projects (or equivalently, the use of transfers from a single study to many separate instances of the same resource or impact) which collectively have an impact larger than the sum of the parts. Accumulating pressures and impacts on coastal and marine ecosystems are not isolated and independent, but synergistic, with feedbacks and interactions that cause individual effects to be greater than their sums (Waltham *et al.*, 2020). The potential for cumulative impacts should always be considered, and it must be understood that the implementation of one decision may change the benefit and cost functions elsewhere (in space and in time), particularly in the case of large projects.

Non-linearities, risks of moving to regions of highly inelastic demand/rapidly changing values, and threshold effects have implications within individual studies, and for attempts to transfer values across studies, to gross-up across spatial scales, or to construct meta-analysis functions. More generally, they may suggest the need to move to safe minimum standard or precautionary approaches when dealing with decisions about critical natural capital. This may imply setting limits to the applicability of cost-benefit methods where catastrophic and/or irreversible changes are a possibility. Decisions to protect, conserve, or delay development are generally reversible, in the sense that the development or conversion option will probably remain open in future. This leads to an asymmetry that is reflected in economic value frameworks by the concept of (quasi-)option value. That said, resources available for conservation are scarce, meaning that resources are insufficient to fund all desirable conservation activities, and investing in less beneficial projects, or in projects that do not achieve restoration success, may represent an irreversible lost opportunity to prevent biodiversity loss.

4. APPLICATIONS TO MARINE CONSERVATION

There is a substantial literature on valuing the costs and benefits of marine resources, services, conservation, protected areas, and restoration (Mehvar *et al.*, 2018; Vassilopoulos and Koundouri, 2017; Lipton *et al.*, 2014) although relatively little looking at both costs and benefits together or applying a full cost-benefit approach. Reviews and meta-analyses exist for specific habitats and services, for example Rao *et al.* (2015) on shoreline protection values, Hynes *et al.* (2018) on marine recreation, Laurans *et al.* (2013) on coral reef services. Littles *et al.* (2018) focus on identifying the beneficiaries of coastal services. Numerous databases have also been compiled containing marine ecosystem service value estimates. For example, the Marine Ecosystem Services Partnership (MESP) database holds over 1000 entries of economic valuation data representing over 2000 values. Others include the US National Ocean Economics Program (NOEP) database and TEEB Ecosystem Services Valuation Database[11].

Nevertheless, a recent review (Milon and Alvarez, 2019) reveals significant gaps in research and understanding of coastal and marine ecosystem services and economic values, and raises concerns about aggregating individual ecosystem service values when there is weak understanding of the relationships and feedbacks between ecosystems and the services they produce. Particular gaps exist for the deep sea; indeed Thurber *et al.* (2014) argue that ecosystem services frameworks developed for terrestrial environments may not be suited to the deep sea, due to the low resolution of spatially explicit marine information and the difficulty of quantifying ecosystem functions and processes in the highly dynamic and connected three-dimensional marine environments. Many of the services identified by Armstrong *et al.* (2012) are supporting or intermediate services in the deep sea

[11]For the MESP database see https://marineecosystemservices.org/about/ and for the NOEP database see https://oceaneconomics.org/default.aspx

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that underpin crucial final services elsewhere in space and time – this does not sit well with recent approaches such as CICES or FEGS which focus only on final services. Le *et al.* (2017) also highlight the likelihood of discovering unknown final and supporting services.

Integration of ES valuation into marine and coastal policy formation is further complicated by the fact that these ecosystems tend to be large and often overlap multiple political jurisdictions and economic sectors, including Areas Beyond National Jurisdiction (Hanley *et al.*, 2015). The complexity of marine ecosystems, and their connectivity with other systems and services across space and time, makes knowledge transfer very challenging (Jobstvogt *et al.*, 2014). Nevertheless, there are several examples of marine/coastal CBA with the potential to inform decisions.

There is an extensive literature on valuation of marine protected areas (Hargreaves-Allen 2020), with cost estimates (*e.g.*, Balmford *et al.*, 2004; Sumaila *et al.*, 2007) and studies of marine reserve benefits (*e.g.*, Russ *et al.*, 2004; Gell and Roberts 2003; Halpern 2003). Some studies focus on specific benefits, *e.g.*, recreation (Paltriguera *et al.*, 2018), angling (Pouso *et al.*, 2020), but there is still relatively little that combines monetary estimates of costs and benefits. Quantitative analyses of marginal changes to ecosystem services remains rare, due to lack of data and complex modelling, non-linearities and threshold effects (Hargreaves-Allen 2020). However, there are exceptions that demonstrate the potential for wider use of these methods.

CBA provides a rational and methodical approach to structuring and presenting information that can be useful even where most of the impacts cannot be expressed in monetary terms. Sumaila *et al.* (2007) apply a CBA framework to argue for marine reserves in the high seas, estimating US$270m annual profit loss from 20% closure of high seas fisheries, contrasted with US$152m annual subsidies to high seas deep-sea bottom trawlers alone. Longer term benefits, including fishery gains and reduced risks, are discussed but not quantified, highlighting key data gaps that could be addressed to draw firmer recommendations. Waldron *et al.* (2020) model 30% marine protection, showing it produces an initial shock then restores growth to fisheries, whereas without expanded protection fisheries contract in the mid-term.

A similar strategic approach can be combined with expert judgement and extensive sensitivity analysis to address data gaps. The Impact Assessment (Defra 2010) for the Marine Conservation Zone (MCZ) provisions in the UK Marine and Coastal Access Bill draws on several studies (ABPmer *et al.*, 2007; SAC and University of Liverpool, 2007; Hussain *et al.*, 2010;) to conduct a CBA of marine protected areas in the UK, at a strategic/national scale. The study identifies eleven ecosystem service impacts and values seven of these[12]. Because there is evidence of the total value of these services, but limited evidence of the impact on services of a specific policy change, expert judgement is used to score likely impacts; to be conservative, only on-site effects are considered. The NPV over 20 years is estimated as £13.0bn (£7.9bn-£18.0bn) on best estimates, with BCR 6.7-38.9, depending on scenario. Sensitivity analysis suggests a positive NPV is robust even to worst case scenarios that significantly reduce benefits and increase costs. This conclusion is strengthened by non-use values, estimated as £487-£1200m per annum through a separate stated preference survey (McVittie and Moran 2010), but omitted from the CBA to avoid any risk of double counting. The overall outcome allows confidence that the proposals would bring net social benefits, although the precise level of the benefits remains highly uncertain.

Partial cost-benefit can sometimes be sufficient with only market values. Hodgson and Dixon (2000) present CBA of halting deforestation in a watershed area to prevent sedimentation of downstream coral reefs, showing that the costs of foregoing income from logging would be much less than the benefits of preserving the fishing and tourism industries. Elsewhere, Homarus Ltd. (2007) considers a proposed conservation zone of 60 square nautical miles centred on Lyme Regis (UK), stopping destructive scallop dredging but allowing more sustainable forms of fishing (dive catching of scallops, crustacean potting and fixed netting of skates and rays) and recreational use. Since market returns from the protection exceed those from the business-as-usual case, and environmental benefits are unknown but certainly positive, this established a strong case for protection. However, it is possible that including non-market benefits could justify stronger protection, and/or a larger conservation area.

Other studies go beyond the focus on market returns to include some more easily quantifiable ecosystem service impacts. This is becoming a common approach for projects with strong carbon implications (*e.g.*, Sasaki and Yoshimoto, 2010) since carbon valuation can be strongly tied to policy priorities and a basis for pricing can be found in existing carbon markets or abatement cost estimates (*e.g.*, BEIS, 2021). The argument is essentially that economic impacts plus the value of carbon changes are themselves enough to justify a project, and in addition there are

[12] Additional carbon savings (not related to ecosystem services) are also considered.

other, non-monetised ecosystem service benefits that are unequivocally positive and therefore can only strengthen the result.

Several reviews are available on the costs of marine ecosystem restoration (Spurgeon, 1999; Bayraktarov *et al.*, 2016, 2019). Papadopoulou *et al.* (2017) review over 400 sources on restoration and present a review of success and failure factors and costs and benefits for key marine habitats. These reviews demonstrate that the costs and benefits of restoring coastal and marine ecosystems can vary substantially, depending on the technique, the habitat, and the scale of the operation. Many studies are experimental and small scale, so it is difficult to extrapolate estimates to CBA of wide-scale restoration strategies where economies of scale could be important.

Restoration can involve a complex suite of passive and active measures that can complicate analysis. For example the Chesapeake Bay Program Partnership (formed in 1983) estimated the economic benefits of cleaning up the watershed at US$130 billion annually (Stewart-Sinclair *et al.*, 2020). Seagrass recovery was slow in the bay and has been aided by seed-based restoration in plots from 100m^2 to 200 ha across four coastal bays (Orth *et al.*, 2012), with seed dispersal and reproduction spreading restored seagrass to approximately 2500 ha of seafloor (Stewart-Sinclair *et al.*, 2020). The benefit assessment has been estimated at about $1.3bn to $7bn per year, with an additional $0.7bn-$1.1bn capitalised in enhanced waterfront property values (Wainger *et al.*, 2017). Around 80% of the annual benefits are for non-use values. The values also include co-benefits of the control measures, not related to estuarine water quality, including health, safety, climate risk reduction, and hunting opportunities. The value of improved resilience of systems/services is as a key omission: Wainger *et al.* (2017) argue that WTP for more reliable services could be estimated but changed probabilities of a system reaching a tipping point are unknown, so expected values could not be calculated. They propose quantitative, non-monetary metrics to indicate level of benefits that could be considered alongside the CBA.

Partial CBA of specific interventions has been possible, notably for large-scale oyster restoration. Blomberg *et al.* (2018) report that from 2000–2011, more than $45m was invested in 187 projects to restore over 150ha of oyster reef habitat, primarily in the Chesapeake Bay area and Florida Gulf coast. Trends over time indicate that projects are being implemented at larger scales, increasing from an average of less than 0.4ha in 2000 to over 1ha on average in 2011. Costs per unit decreased from an average of more than $2.1 million per ha in 2000 to just over $500,000 per ha in 2011.

In Maryland, Stewart-Sinclair *et al.* (2020) report US$51 million invested in 298ha of oyster restoration resulting in annual benefits of US$22.3m in fisheries, 313 jobs, and US$3-18m nitrogen removal (the cost of $171,000ha^{-1}$ is much cheaper than the Blomberg *et al.* estimates). Blomberg *et al.* report that lack of monitoring data and assessments of success is a major problem in restoration ecology. Griffin (2016) presents CBA of oyster restoration in Rhode Island, where restoration costs could be recouped through ecosystems services in 17 years, but in fact NPV is negative due to the need for repeated restoration interventions – highlighting the importance of site selection and removal of pressures before attempting restoration. Lester *et al.* (2020) report that poor siting decisions can contribute to failed restoration projects, and that low success rates result in high average cost estimates. Even where the problem is restricted to site selection, considering costs and wider benefits can be materially important to success (Ando and Langpap, 2018).

Weber (2015) applies value transfer from a review of 29 estimates of values for pacific salmon restoration in the US. Despite a wide range of values, any of the estimates, aggregated over time and local population, supports substantial recovery efforts for their case (Willamette Spring Chinook). Weber (2015) notes that while CBA findings may not appear relevant for a species that is already listed under the Endangered Species Act, "the decision space for recovery is broad and estimates of TEV can inform policy decisions." This is because economic criteria are taken into account[13] for designating critical habitat designation. Since 'disproportionate costs' provisions are common throughout environmental policy, better benefits estimates can be important even where protection appears to be strict.

Batker *et al.* (2005) ostensibly focus on restoration of only 2 acres at North Winds Weir in the Green/Duwamish and Central Puget Sound Watershed. However, the resulting "transition zone habitat" in the short zone where freshwater meets tidal salt water (5.5-7 miles from the river mouth) is so scarce and essential to salmon that extirpation could occur without it. Hence although the present value of local benefits ($384,000-$1.36m) are insufficient to justify the $3.69m land acquisition and restoration costs, the off-site benefits of protecting salmon for the river system are estimated at $19m per acre. This is akin to designating a sufficient area of transition zone habitat as critical natural capital, and the question becomes one of deciding the least-cost place to locate it.

[13] Under 16 U.S.C. § 1533(b)(2).

CBA has been widely applied in coastal zone restorations aimed at flood control through managed realignment and habitat creation. Generally, these are based on value transfer for specific projects (*e.g.*, Tinch and Ledoux 2006; Tinch and Provins 2007; Everard 2009) while some have used original valuations and strategic scenario analysis at estuary scale (*e.g.*, Luisetti *et al.*, 2011, 2014). There is growing awareness that 'soft' engineering using 'green infrastructure' /nature-based solutions can offer win-win solutions compared to traditional 'hard' structures, enabling effective flood and erosion management, often at lower cost, while simultaneously achieving conservation goals, leading to strong positive NPVs (eftec 2015; Deely *et al.*,2020). Results are highly location-specific, in particular because the flood protection benefits are heavily dependent on the value of assets protected, but also due to variations in local ecology, human populations and environmental conditions. Da Silva *et al.* (2014) use value transfer to estimate £1,875-£3,500$ha^{-1}y^{-1}$ net service change for creating 262ha of intertidal habitat at the Steart Peninsula (UK) while MacDonald *et al.* (2017) use the TESSA toolkit to estimate net service change of £1,460 $ha^{-1}y^{-1}$ for 180ha in the Ribble Estuary (UK) and £575$ha^{-1}y^{-1}$ for 162ha in the Firth of Forth (Scotland). Boerema *et al.* (2016) demonstrate the importance of accounting for ecological succession, with annual values of restored tidal marsh in the Schelde estuary (BE/NL) varying from €20,000 to €80,000 $ha^{-1}y^{-1}$ depending on the successional stage. These much higher values are explained in part by inclusion of significant values for nitrogen removal, omitted from the other studies. In the UK, transfer studies are now facilitated by guidance on natural flood management appraisal following a suite of studies by the Environment Agency.[14] Davis *et al.* (2019) present a generalisable natural capital valuation method for prioritising managed realignment investments, taking account of opportunity costs to agricultural production, direct re-alignment costs, property damages (avoided), carbon sequestration benefits and recreational benefits. The scope for restoration as part of flood and coastal erosion risk management is substantial. Vousdoukas *et al.* (2020) estimate that costs of dike-raising outweigh benefits for 67-89% of the European coastline, depending on scenario. Natural processes, perhaps managed, are likely to dominate in those areas, and may also be used instead of or alongside hard structures elsewhere.

There will also be spatial variation in the value of ecosystem services that is not related to the supply-side determined by ecological and biophysical aspects, but rather to demand-side features of human populations and preferences (Tallis *et al.*, 2011). Nevertheless, Lester *et al.* (2020) review 572 papers on restoration and

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[14] https://www.gov.uk/government/publications/working-with-natural-processes-to-reduce-flood-risk

report that fewer than 5% examined site selection or applied spatial planning principles; of those, almost all focused on site selection, but not on scale or configuration. Research has focused more on alternative restoration techniques within a single site than on where to conduct restoration in the first place. This may be because restoration has often been reactive/opportunistic more than planned, and this could change under the current policy agenda. But despite the wide array of spatial planning tools, decisions about where to target restoration are not yet sufficiently grounded in spatial analyses that explicitly consider alternative sites, spatial scales and the values of ecosystem service changes.

5. DISCUSSION AND CONCLUSIONS

Valuation is not essential: there are alternative ways of carrying out appraisal (*e.g.* MCA, collective decision methods) and even environmental taxation could be implemented without using valuation to set the tax rates. But does valuation make these processes easier, more defensible, more transparent, more (cost-) effective? Are arguments for protecting and restoring ecosystems more convincing, for some decision makers or in some contexts, if they're expressed in monetary value terms? Does valuation evidence help decision makers to take full account of environmental factors, and does this result in better decisions about trade-offs? The extent to which valuation is useful will be dependent on environmental, economic, and social/political contexts, and there will always be bounds on the appropriate uses of values. The key issue is not whether monetary valuation is 'accurate', 'complete' or 'true', but rather to determine the conditions under which monetary valuation may be useful, and the risks of worsening outcomes or decisions due to using – or not using – valuation in any given context.

Firstly, we should recognise that there are many different purposes and interpretations of valuation and appraisal. Specific project appraisal is most familiar, but the methods are also applied for strategic scenario analysis, for communication purposes, demonstrating value for money, prioritising investments with scarce funds, and so on. Each of these may have different requirements for accuracy and research expenditure commensurate with the context and the audience for the results. Similarly to other areas of science, good quality CBA can be valid and useful, while low quality CBA is of little help or even misleading. But 'good quality' does not mean that all costs and benefits must be valued. On the contrary, it is better to value only that which can be valued with reasonable confidence, within the scope and bounds of the objectives of the study. What is 'reasonable' can take into account the potential for sensitivity analysis to reveal threshold values for specific service changes, and the potential value of exploratory/tentative valuations, provided these uncertainties are spelled out clearly. Non-valued changes should be identified clearly, where possible in quantitative terms, and, failing that, qualitatively. This provides a more useful input to decision support than a set of numbers lacking in accuracy or context. Even a CBA that seems to have many omissions, from the strict perspective of covering the full impacts of projects and interventions, may nonetheless be a practical and useful study within the context of a specific problem or decision context.

To make good decisions, it is important to understand the potential economic, social, and environmental impacts, benefits and costs of protection, restoration and exploitation actions. This requires a broad strategic view of the marine space and its role in achieving sustainable development, including in the context of climate change adaptation and mitigation. We need to recognise the full range of values arising from marine environments, including roles in vital biogeochemical cycles and conservation values. Finally, we must also understand the different motivations and incentives faced by different actors involved in managing, exploiting, conserving and restoring marine environments, and recognise the need for policy structures to ensure that private decisions are consistent with socially desirable outcomes.

Globally, however, there is little evidence that the growing body of valuation evidence is being used in the management of marine resources (Torres and Hanley, 2017). Ruckelhaus *et al.* (2015) describe the incorporation of valuation into decision processes as "painstakingly slow". Significant barriers include lack of scientific knowledge of key ecosystem service linkages, lack of relevant economic valuation studies, methodological problems applying certain valuation methods to marine issues, and lack of public familiarity with marine ecosystems (Hanley *et al.*, 2015). Marre *et al.* (2016) survey Australian coastal/marine decision-makers, and find that a large majority are familiar with economic valuation of ecosystem services, and consider it useful or necessary for decision-making, but never or rarely use it. Nyborg (2012) argues that CBA results are only included in political decision making when they support the preferred political outcome. The VALUES project[15] reported that assessments are often commissioned, designed and conducted "in ways that do not achieve their full potential in terms of practical usefulness and policy relevance", in part due to a failure to balance "the trio of credibility, legitimacy and relevance", including weak links from assessment processes to public and private policy-making." (Berghöfer *et al.*, 2016). Similarly, the European Court of Auditors (2019) found several failings in the Commission's

[15] http://aboutvalues.net/

implementation of environmental accounting, reducing usefulness for policy, including failure to establish a comprehensive action plan or a long-term view of data needs and indicators for policy support. Milon and Alvarez (2019) review studies on use of economic valuation information in coastal/marine planning and policy in the EU, US, Caribbean and Australia, arguing that valuation information is not widely understood and has had negligible impact on policy processes, but suggest that "a more encompassing framework" such as wealth accounting could help.

Nevertheless, Hooper *et al.* (2019) report that existing frameworks, in particular the classification of ecosystem services and the cascade from ecological assets to benefits and values, are broadly fit for purpose. The most significant gaps are in understanding how ecosystems support the delivery of services, and in empirical valuation data, where there are few estimates for regulating services and some cultural services and the lack of high quality original studies limits the scope for defensible value transfer. Fonner *et al.* (2020) argue that growing recognition of the need to allocate scarce conservation resources effectively, and of the improved ecological outcomes when taking economic factors into account, is building momentum for greater use of valuation. Key challenges include capacity building to integrate ecosystem services and valuation information more effectively in policy making, including better understanding of policy needs among the research community (Sitas *et al.*, 2014). Developing the political will and financial backing to achieve the goals of the EUBS2030 will require strong valuation and appraisal evidence to build business cases, leverage financing, overcome resistance in communities more focused on the social and economic objectives, and ultimately to achieve the long-sought mainstreaming of biodiversity and environmental concerns across all policy sectors.

The EUBS2030 section (3.3.3) on "Measuring and integrating the value of nature" recognises that "Biodiversity considerations need to be better integrated into public and business decision-making at all levels" then continues with the promise that, building on the existing work (notably MAES and KIP-INCA) "the Commission will develop in 2021 methods, criteria and standards to describe the essential features of biodiversity, its services, values, and sustainable use". Methods cited are environmental footprints, life-cycle approaches, and natural capital accounting, but there is no direct mention of valuation beyond that. There may be a risk that the focus on natural capital accounting, and more generally on green/blue growth and market instruments, could create a focus on exchange values at the expense of welfare values required for other purposes such as policy appraisal. This would be a regressive step insofar as representing the actual values to people and improving environmental justice are concerned.

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One of the first EUBS2030 actions is for the Commission to "put forward a proposal for legally binding EU nature restoration targets" (with priority for capturing/storing carbon and preventing/reducing impacts of natural disasters). This proposal will be subject to an impact assessment including the possibility of an EU-wide methodology to map, assess and achieve good condition of ecosystems (Maes *et al.*, 2020). This will seek to identify both the conditions in which the targets must be met and the most effective measures to reach them. For these purposes, and for individual appraisals of restoration projects and plans, it is important that the monetary values used should be welfare values wherever possible, although there are circumstances in which proxies (based for example on avoided costs) can be acceptable. It is clear from review of the literature that some of the most important values attached to ecosystems generally, and to restoration of nature in particular, relate to cultural services. These include some services that can be valued in either exchange or welfare terms, for example outdoor recreation, and others for which exchange values may be essentially non-existent, for example non-use values. In both cases, however, inclusion of consumer surplus could make a material difference to the outcomes of appraisals. This applies to all forms of restoration, including passive and active approaches.

Any form of restoration requires dealing with the pressures that have caused degradation, and in most cases these are economic activities that need to be modified in order to reduce impacts. There will be some win-win solutions, but in other cases changes will entail economic costs, and it will be important to demonstrate that the restoration/conservation benefits, including improvements in ecosystem services, exceed those costs. Often this will involve other benefits not directly associated with the restored system per se. For example pollution reduction may be an essential precondition for restoring seagrass/kelp but will also mean improved bathing water quality and other benefits. Appraisal needs to take account of this by identifying and wherever possible valuing those benefits.

It is often also useful to consider the value of information that might be gleaned from an activity, and associated monitoring and assessment, for example in terms of improved restoration techniques as well as better knowledge of how ecosystems and services react to interventions. There is a growing empirical literature on the value of information for marine resources management (Essington *et al.*, 2018; Hutniczak *et al.*, 2019; Bisack and Magnusson, 2014; Polasky and Solow, 2001). The potential to gain valuable knowledge from experience can be an important component of CBA in its own right (and can be seen as set against the often significant costs of monitoring or designing/implementing adaptive management regimes)

Specifically for marine systems, the EUBS2030 highlights that the "need for stronger action is all the more acute as marine and coastal ecosystem biodiversity loss is severely exacerbated by global warming." This includes the problems of sea-level rise and coastal squeeze, and in this context it should be noted that 'restoration' may mean helping intertidal systems to migrate such that the habitat areas and the services provided by these systems are maintained or restored. The Strategy notes the importance of restoration of carbon-rich ecosystems as well as important fish spawning and nursery areas. Carbon-rich ecosystems can have a crucial role to play in climate policy (mitigation and adaptation), so it is important to measure and value these services.

Carbon valuation is one of the most feasible services to value, and links closely to the Green Deal objectives. However, there remain debates on precisely which values to use – in particular, whether valuation should focus on damage costs, prices from carbon markets, or abatement costs consistent with reaching targets (as in the UK). Given the Green Deal objective of reaching climate neutrality by 2050, there is an opportunity for linking the valuation of carbon sequestration to the marginal abatement costs associated with that policy target. The choice could make material differences to the outcomes of appraisals and should be addressed in the development of methods, criteria and standards.

The methods should also reflect that ecosystem service values could be useful as baselines and indicators of ecosystem quality. This is especially important in the context of global change, because coastlines and species ranges are shifting, and the definitions of "restoration" and "good quality" will need to be sensitive to this. In such cases it is not possible simply to consider what an area was like in its 'pristine' past.

The development of methods should also seek to advance in line with other initiatives, not only the SEEA-EA which is focused on exchange values, but also processes including the IPBES Values Assessment, and the standards ISO14007 and ISO14008. It should be noted that both ISO standards focus on welfare values, taking "an anthropocentric perspective" that "includes use and non-use values as reflected in the concept of total economic value when environmental costs and benefits are determined in monetary terms."

Methods should also acknowledge the problems identified above relating to failure to integrate value assessment with policy processes. Several important limits are already well-recognised in economics, notably the over-riding need to protect critical natural capital, and more generally recognition that values change with quantities. Decision support tools are not operated as independent calculations but are applied within the context of a set of broader governance principles that set down the goals of government/society and the procedures and modus operandi that are considered legitimate and appropriate. Principles endorsed by many states and enshrined in various legal structures and international agreements include the Precautionary Principle and the Polluter Pays Principle. Legitimacy also requires consultation over the trade-offs and uncertainties involved in resource management. Decision support tools are ways of structuring the available information to help decision-makers to understand the trade-offs involved in decisions; they do not provide 'final answers' or replace the need for deliberation. This caveat applies to all applications, but *a fortiori* where there is high uncertainty regarding impacts, or a risk of irreversible or catastrophic outcomes.

This is recognised in EU policy, for example the EUBS2030 proposes that the EU "should advocate that marine minerals in the international seabed area cannot be exploited before the effects of deep-sea mining on the marine environment, biodiversity and human activities have been sufficiently researched, the risks are understood and the technologies and operational practices are able to demonstrate no serious harm to the environment, in line with the precautionary principle and taking into account the call of the European Parliament". Under the MSFD, Good Environmental Status threshold values should be set "on the basis of the precautionary principle, reflecting the potential risks to the marine environment" (EC, 2017). But beyond that, there remains a large margin for manoeuvre, and a pressing need for integration of environmental values in policy processes.

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