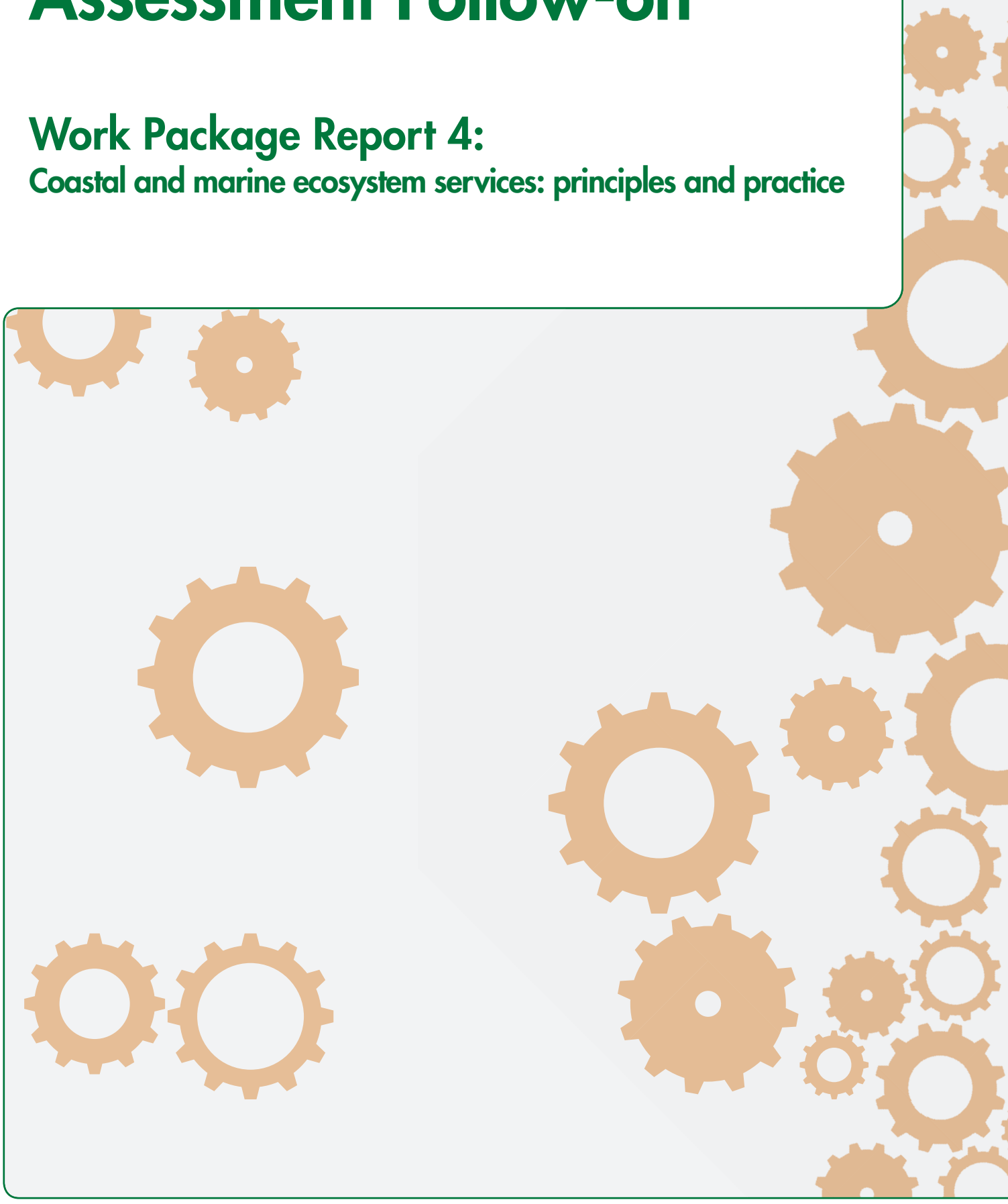
UK National Ecosystem Assessment Follow-on

Work Package Report 4:

Coastal and marine ecosystem services: principles and practice



Principle Investigators: Kerry Turner[1], Mike Elliott[2], Laurence Mee[3]

Authors: Kerry Turner[1]*, Marije Schaafsma[1], Mike Elliott[2], Daryl Burdon[2], Jonathan Atkins[4], Tim Jickells[5], Paul Tett[3], Laurence Mee[3], Sonja van Leeuwen[6], Steve Barnard[2], Tiziana Luisetti[6], Lucille Paltriguera[6], Gianna Palmieri[1], Julian Andrews[1]

Report Review Editor: Eva Roth[7]

Affiliations:
[1] CSERGE, School of Environmental Sciences, University of East Anglia, Norwich, NR4 7TJ, UK
[2] Institute of Estuarine & Coastal Studies, University of Hull, Hull, HU6 7RX, UK
[3] Scottish Association for Marine Sciences (SAMS), Oban, Argyll, PA37 1QA
[4] Hull University Business School, University of Hull, Hull, HU6 7RX, UK
[5] School of Environmental Sciences, University of East Anglia, Norwich, NR4 7TJ, UK
[6] Cefas, Pakefield Road, Lowestoft, NR33 0HT, UK
[7]University of Southern Denmark, Campusvej 55, 5230 Odense M, Denmark

*Correspondence to: Kerry Turner, r.k.turner@uea.ac.uk

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Contents

Abbreviations and acronyms

AM	Adaptive management
AON	Apparently On Nest
BAU	Business as usual
BBN	Bayesian belief network
BPEO	Best practicable environmental option
BT	Benefit transfer
CBA	Cost benefit analysis
CCW	Countryside Council for Wales (now Natural Resources Wales)
CE	Choice experiment
CEA	Cost effectiveness analysis
CFP	Common fisheries policy
CI	Confidence interval
CSTT	Comprehensive studies task team
CV	Contingent valuation
DDR	Declining discount rate
DPSI(W)R	Drivers, Pressures, State changes, Impacts (Welfare) and policy Response
DSS	Decision support system
EcoQO	Ecological quality objective
ESF	Ecosystem services framework
EEZ	Exclusive economic zone
ERSEM	European regional seas ecosystem model
GCM	Global coupled model
GEcS	Good ecological status
GEnS	Good environmental status
GDP	Gross domestic product
GHG	Greenhouse gases
GIS	Geographic information system
GVA	Gross value added
IPCC	Intergovernmental Panel on Climate Change
LLGHG	Long-lived greenhouse gases
MCA	Multi-criteria analysis
MNR	Marine nature reserve
MPA	Marine protected area
MPS	Marine Policy Statement
MR	Managed realignment
MSFD	Marine strategy framework directive
NGO	Non-governmental organisation
NTZ	No Take Zone
PPE	Perturbed physics ensemble
QOV	Quasi-option value
SAF	Systems approach framework
SCC	Social cost of carbon
SES	Social-ecological system
SMART	Specific; measurable; achievable / appropriate / attainable; realistic / results focussed / relevant; and time-bounded / timely

SP	Stated preference
SRES	Special Report on Emissions Scenarios
TC	Travel cost
TEEB	The Economics of Ecosystems and Biodiversity
TEV	Total economic value
TSV	Total systems value
UKBAP	UK Biodiversity action plan
UK NEA	UK National Ecosystem Assessment
UK NEAFO	UK National Ecosystem Assessment Follow-on
VNN	Valuing nature network
WFD	Water framework directive
WTP	Willingness to pay

Key findings

Understanding and adaptively managing the impacts of the diverse and dynamic environmental change experienced in coastal zones requires better interdisciplinary knowledge, methods and tools. Adaptive coastal management requires a flexible decision support system in order to enable actual changes in policy and management practice, and follow-up through 'learning by doing'.

The UK NEA Ecosystem Services Framework has been adapted by the UK NEAFO for the UK coastal and marine environment in order to identify its specific components and processes, intermediate and final ecosystem services, and goods and benefits. This Framework and its related tools provide a pluralistic foundation for the use of adaptive management principles in UK coastal policy practice.

An expert-based scenario analysis by the UK NEAFO suggests that UK marine ecosystems would prove resilient to temporary shocks, and that there will be some improvement in ecosystem services as a consequence of present trends in environmental policy. Experts were asked to assess how marine ecosystem services would alter by 2060 under five scenarios, and how these services would respond to shocks, such as shading by volcanic dust for half a year, pollution as a result of the Thames Barrier overtopping, or financial crisis similar to that of 2008.

The UK NEAFO has developed specific indicators, informed by a drivers, pressures, state changes, welfare impacts and policy responses pressures (DPSWR) scoping framework, for six ecosystem services: fisheries and aquaculture, sea defence, prevention of erosion, carbon sequestration/storage, tourism and nature watching, and education. These multiple indicators are necessary to capture the complexity of the marine system associated with even single ecosystem services. It also detects changes over time in marine ecosystem service provision in relation to management measures. The set of practicable ecosystem indicators was developed to reflect 'state changes' and 'welfare impacts' relating to ecosystem services supply. These indicators meet operational requirements and are grounded within the NEAFO ecosystem service and management frameworks.

The UKNEAFO concludes that one pragmatic way to link terrestrial models for nutrient flows from land use in catchments to models for estuaries and coastal waters to assess ecosystem services provision, is through the use of estuarine box models. A box model is a model without spatial representation, which captures the main dynamics as a function of time and driving pressures. There are different types of models available to assist in the effective management of the range of final ecosystem services and their goods and benefits.

The UK NEAFO argues that the future goal for economic assessments of sustainable coastal management should be to measure and value service flows and changes in stocks (i.e. ecosystem health). A separate and complementary ecosystem services account or index may also be a worthwhile objective.

There are considerable gaps in the current valuations of UK coastal and marine ecosystem services, including those benefits deemed important by experts. More primary valuation studies are needed for reliable social welfare assessment. A review by the UK NEAFO found 208 international studies, of which, 25 provide UK-based value estimates. The main gaps relate to the biodiversity and seascape values (non-use existence values) of the majority of global coastal and marine habitats, and some of the

typical UK habitats, such as machair. Both temporal and cultural bias constraints remain formidable challenges for any benefits transfer exercise.

The UK NEAFO has promoted the Balance Sheet approach as a pragmatic format for collating, interrogating and presenting evidence. It is both a process and a tool which addresses the complexity of real world decision-making and trade-offs. It captures economic, ecological and social/deliberative perspectives in trade-off assessments. This not only incorporates efficiency, but also considers the distribution of gains and losses, resilience and carrying capacity aspects of sustainable management.

Summary

A summary of this report can be found separately.

4.1 Introduction: from science to values and decision making

The UK NEA ecosystem services framework (ESF) and related tools are now under test or are being implemented across UK environmental policy circles (e.g. Saunders *et al.*, 2010 for the Crown Estate, Fletcher *et al.*, 2012 for Natural England). In order to assist the adoption process, a number of flexible 'ground rules' may prove useful in order to guide the application of this ESF framework and related decision support system (DSS), as well as the interpretation of its results by the policy community and society at large. The over-arching **adaptive management** (AM) approach taken here is built on the foundation principles of pluralism, pragmatism and decision making anchored to the precautionary principle. It will therefore be argued that the ESF necessitates an interdisciplinary perspective and will require decision makers to operate under conditions of uncertainty, where in some contexts 'full' information will not be available but urgent, or at least short run, precautionary action is necessary. Application of this strategy to dynamic coastal environments and their management will involve just such uncertain and often highly contested ('wicked') policy contexts. Coastal process and ecosystem changes can therefore only be better understood and **adaptively managed** on the basis of an interdisciplinary 'knowledge' and 'methods /tools' (DSS) capacity.

While a number of definitions of the coastal zone have been proposed, in this chapter and in line with IGBP LOICZ* (Crossland *et al.*, 2005) the UK's coastal zone can be typically defined as a long narrow feature consisting of mainland, islands and adjacent seas, denoting the zone of transition between land and the marine domain. From a management perspective coasts are affected by environmental changes across a range of temporal and spatial scales including the continuum from river catchment to coastal ocean. In practical terms, the definitions of the coastal zone need to vary according to the type of problem or set of issues being addressed, the prevailing governance regime and the objectives of the management regime (Whitfield & Elliot, 2011). We will focus primarily on coastal systems but with due note given to the interrelationships with terrestrial and deep ocean systems (Mee, 2012).

The rest of the report is organised in the following way:

- Section 4.2: Conceptual framework including ESF and DSS for ecosystem services, within an adaptive management strategy for coastal and marine areas;
- Section 4.3: Marine futures scenarios;
- Section 4.4: Indicators for changes in ecosystem services provision;
- Section 4.5: Modelling environmental change in coastal/marines environments;
- Section 4.6: Valuation of ecosystem services benefits; and
- Section 4.7: Socio-economic appraisal formats.

4.1.1 Scope of Work Package 4

The scope of this Work Package was to adapt the NEA conceptual framework in order to adequately characterise, for marine/coastal systems, a set of relevant ecosystem services and values. This required a better conceptualisation of ecosystem stock and flow positions and value representation. To cope with the inherent uncertainties surrounding environmental change in coastal areas, an adaptive management strategy was defined and buttressed with the ESF and a practical DSS to enable economic and social appraisal and trade-off analysis. The DSS toolbox is comprised of a problem scoping method (the drivers-pressures-state changes-impacts (welfare)-policy response (DPSWR)); futures scenario analysis; ecosystem services change indicators; formal modelling; ecosystem services benefits valuation (monetary and non-monetary); and appraisal/trade-off analysis formats (the 'balance sheets' approach).

4.1.2 Links to other Work Packages

The distinction and analysis of the ecosystem services stock and flow positions makes clear links to WP 1 Report (natural asset check) and WP 2 Report (macro-economic significance). The marine/coastal modelling review had as one of its main objectives the prospects for linking terrestrial land-use change models to estuarine and coastal water environments. This is the primary link to WP 3. The adapted ecosystem services framework for marine/coastal areas includes consideration of both individual and 'shared' ecosystem services values and is therefore linked to both WP 5 Report and WP 6 Report. Marine scenarios were extensions of some of the NEA scenarios and possible responses to link to WP 7 Report and WP 8 Report. The decision support system for adaptive coastal management advocated contains a number of relevant 'tools', including formal and informal models, and so there is a link to WP 10 Report.

4.2 Adaptive management and ecosystem services: conceptual framework

The coastal management framework set out below is hierarchically arranged. It begins with an explanation of the adaptive management strategy and its high level principles. These are used as guidelines for the deployment of the UK NEA (2011) **ecosystem services framework** (ESF) which in turn provides the focus for a practical DSS (a process and asset of enabling tools), the components of which form the basis for economic and social appraisal/trade-off analysis.
Section 4.2 is organised into the following sub-sections:

- a characterisation of the strategic-level adaptive management approach encompassing the NEA ESF and the links to relevant decision support process, tools and methods necessary for more integrated coastal management;
- a classification of coastal/ marine ecosystem services, the stock and flow position and the distinction between intermediate and final services;
- the links between biogeochemical processes, ecosystem services and the goods and benefits they provide to human society with wellbeing/welfare consequences; and
- an outline of the necessary DSS and its components for practical management.

4.2.1 Policy context

The core aim of this chapter is to develop a conceptual framework guided by adaptive management (AM) principles and incorporating the ESF for a DSS, that will foster interdisciplinary research and contribute to a more sustainable management of our coastal zones, while *inter alia* at least maintaining the provision of a set of ecosystem services over time. It will also contribute to the UK adoption of the EU Marine Strategy Framework Directive (MSFD) and will draw lessons from the implementation of the EU Water Framework Directive (WFD) and other related Directives and policies, such as the Common Fisheries Policy (CFP). In the UK, the regional marine planning agenda is now the focus of much policy attention driven by legislation such as the UK Marine and Coastal Access Act (2009) and Marine (Scotland) Act 2010, guided by the Marine Policy Statement (MPS) and operationalised by Marine Plans, which set out how the MPS will be implemented in specific areas. The conceptual approach will build on that formulated by the UK National Ecosystem Assessment (UK NEA, 2011; Balmford *et al.*, 2011; Bateman *et al.*, 2011b) (see **Figure 4.1**.), and will be suitably adapted to the coastal zone context. The UK NEA 2011 focused on the processes that link human society and wellbeing to the natural environment and *inter alia* on the key role ecosystems play in delivering a diverse set of services which directly and indirectly underpin economic progress and human wellbeing.

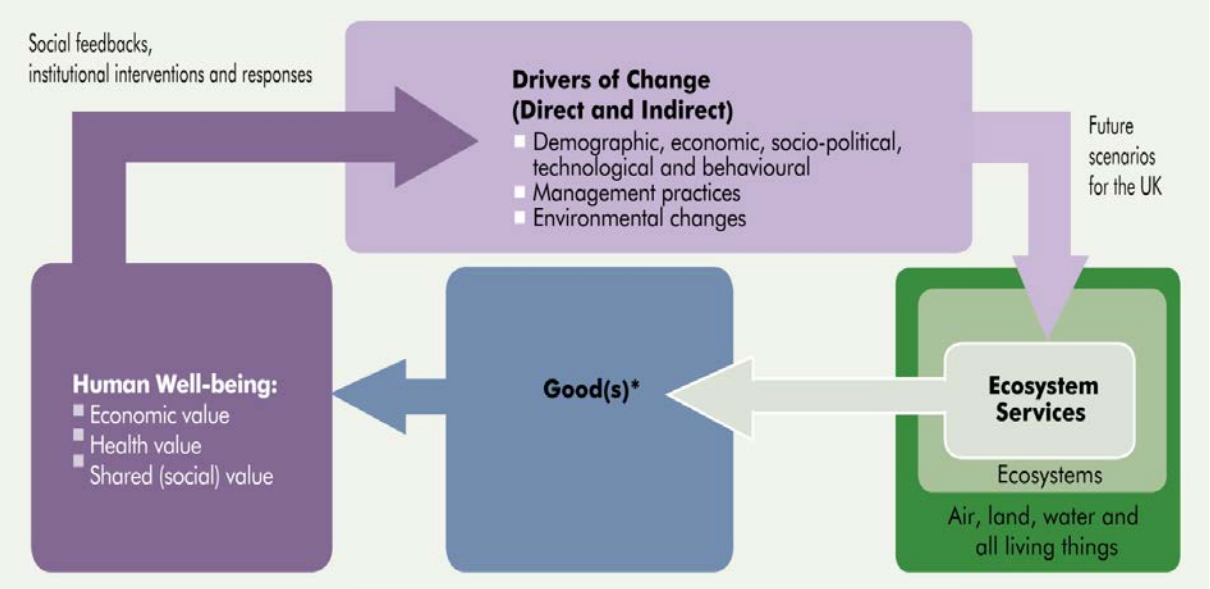

Figure 4.1. UK NEA (2011) Conceptual Framework.

The UK NEA FO (2012-2013) seeks to build on the conceptual and empirical platform for the **ESF** laid down in phase 1 (**Figure 4.2.**). The strategic goal is to build a robust evidence-based case for the embedding of the ESF into the policy process and the workings of the wider contemporary society. However, to foster such a policy switch in practice, new and existing policy tools will need to be combined in a DSS process.

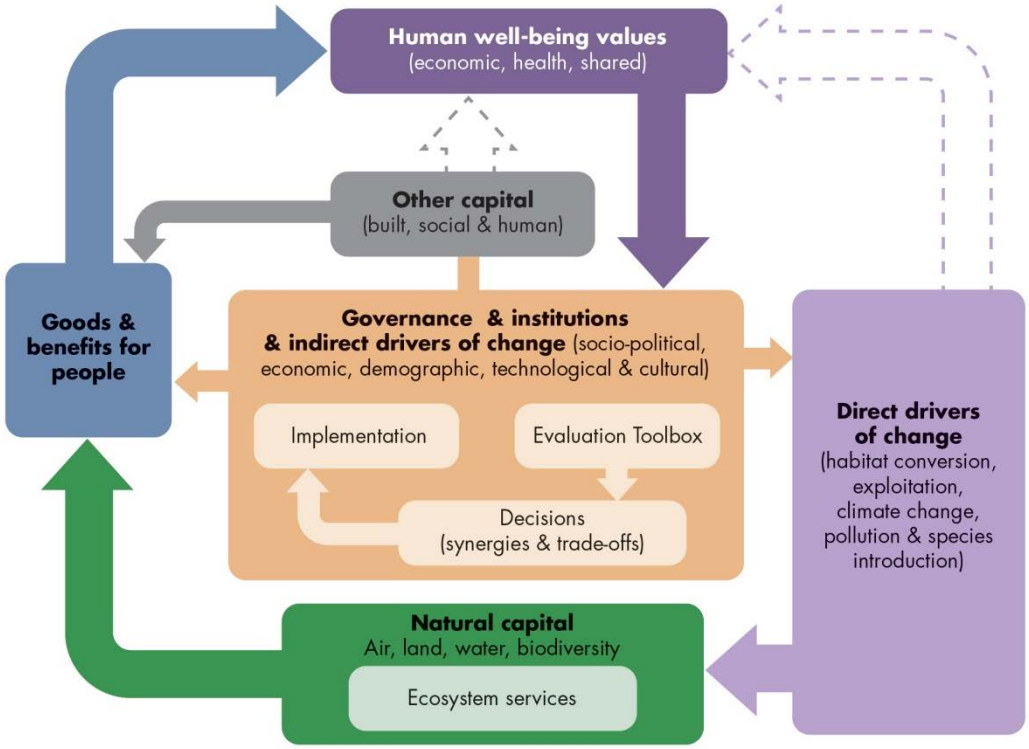

Figure 4.2. UK NEAFO Conceptual Framework.

The achievement of the strategic goals of AM will contribute to a better assessment of the value and significance of the flow of ecosystem services over time, as well as an indication of the stock accounting price or value position (natural asset check) at any given point in time. Genuine economic progress cannot be sustainably achieved without good environmental husbandry principles and practice. Sustainability principles can be used to guide the ESF. This combined approach can then contribute to a fuller quantification and recognition of the true 'comprehensive wealth' of the UK (Gross domestic product (GDP) plus) and how it is changing over time (UNU-IHDP & UNEP, 2012). It is also targeted at policy objectives, such as the possible future adoption of a 'strong' sustainable development path.

The ESF evolved from an earlier natural science-based analytical approach known as the '**Ecosystem Approach**' as detailed by the 1992 Convention on Biological Diversity (CBD). This advocated a much more comprehensive and integrated approach to environmental management. The next step was to augment the systems-based science by the inclusion of social science and humanities thinking, to link ecosystem functioning and its outcomes to the provision of services (e.g. flood protection, recreation, cultural assets supply and many others) which contribute to human welfare/wellbeing. Hence the underlying aim is not so much to solely maximise environmental/ biodiversity conservation, but rather to manage the rate of change in ecosystems (structure (including species composition) and functioning (as rate processes)) as socio-economic and ecological systems co-evolve through time.

4.2.2 Adaptive coastal management: principles.

Coastal zones are institutional domains with administrative boundaries that can cross regional and national jurisdictions and which are not coincident with the scales and susceptibility of biogeochemical and physical processes (known as the scale mismatch problem). The governance regimes operating across coastal zones therefore face particular challenges. However, political, institutional and coastal management agencies and practices (governance) have so far moved only slowly to encapsulate some core conceptual advances provided by coastal zone 'science' (Mee, 2012). These are:

- a recognition that humans are an integral component of the ecology and functioning of ecosystems;
- the connectivity of a river basin catchment and its receiving coastal waters through to the shelf break is a functional unit for coastal resource assessment and management;
- the ecosystem approach (buttressed by the ecosystem services concept) is required to meet sustainable development goals; and that this will require the adoption of multifunctional rather than single service focused interventions;
- that it is possible to assign monetary values to some ecosystem services once translated to societal benefits and to provide non-monetary evaluation of other (e.g. cultural) services benefits;
- that any new DSS needs to be flexible, allowing refinement and adaptation to changing coastal zone circumstances (such as for example the new focus on marine spatial planning) and governance regimes;
- that some global change impacts (in the absence of radical institutional change at the international governance level) such as temperature change, relative sea-level rise and ocean acidification require a pragmatic adaptive response in advance of long term mitigation and/or compensation;
- that there is an increasing need for novel forms of compensation in cases where mitigation of adverse effects is insufficient and where the compensation can be for the habitat (e.g. create new habitat), for a resource (such as restocking of affected fish and shellfish stocks) and for users (financial compensation) (Elliott *et al.*, 2007); and
- that the role of the citizen and individual is as important as central decision making in driving coastal systems quality (Potts *et al.*, 2011).

These are all formidable challenges and better DSSs are required if they are to be successfully overcome and progress is made towards more adaptive coastal management. The environmental change forces (often global) that dominate the zone pose risks that are sometimes exacerbated by overly narrow and short term planning and intervention measures, implemented without due regard for ecosystem processes. This temporal mismatch problem is highlighted by situations in which the slow response time of natural systems is challenging for political processes where there are expectations of rapid outcomes from policy interventions. The slow response time also has profound implications for coastal management options and strategies, forcing policymakers to think about taking actions now with consequences that stretch out far into the future. Warming of the deep-ocean and sea-level rise related to increased greenhouse gas (GHG) emissions, for example, are very slow processes taking up to 1000 years. About a third of the carbon dioxide emitted today will still be in the atmosphere after 1000 years (Stouffer, 2012). We revisit this timescale problem in the context of policy appraisal and the economic discounting procedure later in this chapter, and in **Appendix 4.1**.

In light of the characteristics of coastal zones and policy contexts the adoption of the so-called 'Adaptive Management' (AM) approach at a strategic level is recommended because of, among other things, its emphasis on flexibility and 'learning by doing' practice. Management agencies should therefore be precautionary, giving high priority to coastal functional diversity and related ecosystem services, as well as the maintenance of the system's resistance and resilience, i.e. its respective ability to cope with and recover from stress and shock (Turner, 2000; Elliott *et al.*, 2007; Elliot, 2011). This is a 'stock' quality ('ecosystem health') issue and one that is currently under-researched. We do not know enough about 'minimum' levels of stock structure, processing and functioning and the type and levels of stress that systems can cope with without regime change. This will in turn require the adoption of a relatively broad scale perspective, in order to understand and potentially manage 'landscape' level ecological processes and relevant socio-economic driving forces more cost effectively (de Jonge *et al.*, 2012). A systems-based approach is required to help cope with the inevitable uncertainty that afflicts coastal management and is the basis for AM (Mee, 2005).

The systems-based approach explicitly recognises that most systems are complex and display inevitable uncertainty in the links between causes and effects. AM is a pragmatic way to achieve national and social-ecological objectives in the face of these high levels of uncertainty. It treats management actions in the coastal and marine system as 'experiments' based on the principle of 'learning by doing'. The MSFD employs this approach through their cycle of target setting, planning, implementation and review of marine strategies (Mee *et al.*, 2008). AM can accommodate 'surprise' events by encouraging approaches that build system resilience to withstand stress and shock and help maintain basic ecosystem functionality (Mee, 2005). AM sets both a long term vision (supported by measurable environmental targets, e.g. Good Ecological Status (GEcS) and Good Environmental Status (GEnS) and their indicator sets respectively in the WFD and MSFD), as well as short term goals for ecosystem improvement (see **Figure 4.3.**). In the case of the MSFD, the long term objectives are supranational (regional sea or EU-wide level), whereas the short-term goals are set through national planning processes and function like 'stepping stones' towards the longer term ones. For 'learning' to occur, it is important that appropriate indicators are formulated and progress towards all targets is monitored carefully and communicated in a transparent manner, allowing objectives and goals to be adjusted from time to time as more information becomes available. The overall vision (GEnS in the case of the MSFD) reflects human values towards the marine environment; the term 'Good' is a human-centric one and the measurement of value is critically important (Mee *et al.*, 2008; Borja *et al.*, 2010).

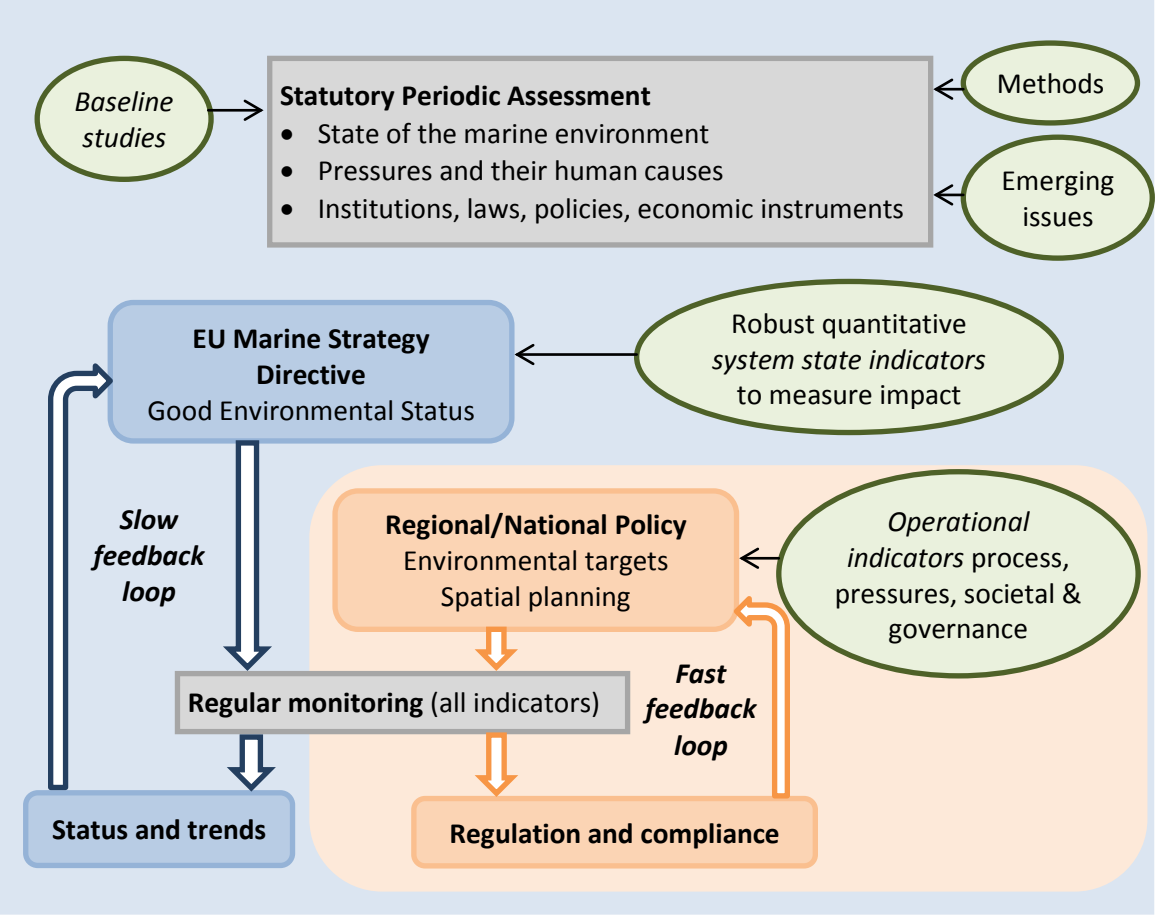

Figure 4.3. Scheme for adaptive management used in current EU marine policy (particularly the MSFD). Source: adapted from Mee (2005).

The linkages between catchment-coastal processes and systems, the influence of climatic change and the impacts on and feedback effects from socio-economic activity all need to be better understood if we are to fully characterise the coastal ecosystem services stocks and flows and assign appropriate values. The incorporation of these data into DSSs, it can be argued, would facilitate better policy outcomes. The values that need to be incorporated are not confined to economic monetary-based values, but encompass a plurality of values expressed in a number of ways, both quantitative and qualitative (Turner, 1999; Chan *et al.*, 2012).

A particular feature of the coastal zone is the so-called 'legacy' problem with 'lock-in' effects and the consequential increased risks and vulnerability to flooding and erosion that it poses. Coastal situations are often conditioned by a historical legacy burden, e.g. the build-up of contaminants in estuarine and coastal sediments from past industrial and urban development; the impact of physical structures and reclamation activities themselves; chronic eutrophication pressures from intensive agriculture or inadequate sewage treatment provision; or depletion of fish stocks by long established fishing practices. This legacy also extends to entrenched historical and cultural use patterns and expectations which may not be environmentally or economically sustainable but can be difficult to alter. Thus the impacts on the stock and flow of ecosystem services can be significant, complex and difficult, and costly to ameliorate, often requiring catchment or wider scale action, combined with continual stakeholder engagement.

Social and economic parameters also change as the process of globalisation continues and its pace of change escalates. Driven by the trends in international trade and finance (and fuelled by, among other factors, persuasive advertising industries) coastal zones are at the forefront of a whole suite of continuously evolving impacts with extensive and significant environmental consequences, e.g. from loss of valuable habitats due to port and navigation channel enlargement and energy resource exploitation, to fishing pressures and tourism over-crowding (Mee, 2012). Given the plethora of drivers across different spatial and temporal scales, any DSS must be anchored to a systematic scoping process and be tempered by a 'learning by doing' management philosophy (Mee, 2005). The ultimate goal is to achieve a sustainable and productive utilisation of the available resource system (stock and ecosystem services flow) and the avoidance of irreversible system changes or collapse with consequent high human welfare losses.

4.2.3 Coastal ecosystems processes and services: stocks and flows

Following the general scheme in **Figure 4.2.**, coastal ecosystem natural capital **stocks** (the ecosystem structure and processes and links to the abiotic environment) possess high biological productivity and provide a diverse set of habitats and species, with a consequent **flow** of ecosystem services (the outcomes from the functioning of ecosystems) of significant **value** (benefits) to human society. From this valuation perspective, a combination of basic processes and 'intermediate' services provide 'final' services of relevance to human welfare ('benefits'). Ecosystem services benefits are the 'exports' from the ecosystem sector to the human economic sector (Banzhaf & Boyd, 2012). The term 'intermediate services' should not be interpreted as signifying lesser significance but rather as a necessary signal that provides technically-correct guidance to avoid double counting when services are valued in economic terms (Fisher *et al.*, 2009). Following the UK NEA (2011) conceptual framework for ecosystem services assessment, the outcomes from the functioning of ecosystems have been generically labelled 'goods' which refer to a range of human welfare benefits derived from the flow of final services provided. But the scope of the delivered final ecosystem services (and therefore the valued goods and benefits) is very wide from food to carbon storage, coastal protection, sea defence,tourism and nature watching (Balmford *et al.*, 2011; Bateman *et al.*, 2011b; UK NEA, 2011). **Figure 4.4.** illustrates the conceptual framework, and a full classification of coastal ecosystem services is shown in **Figure 4.5.**

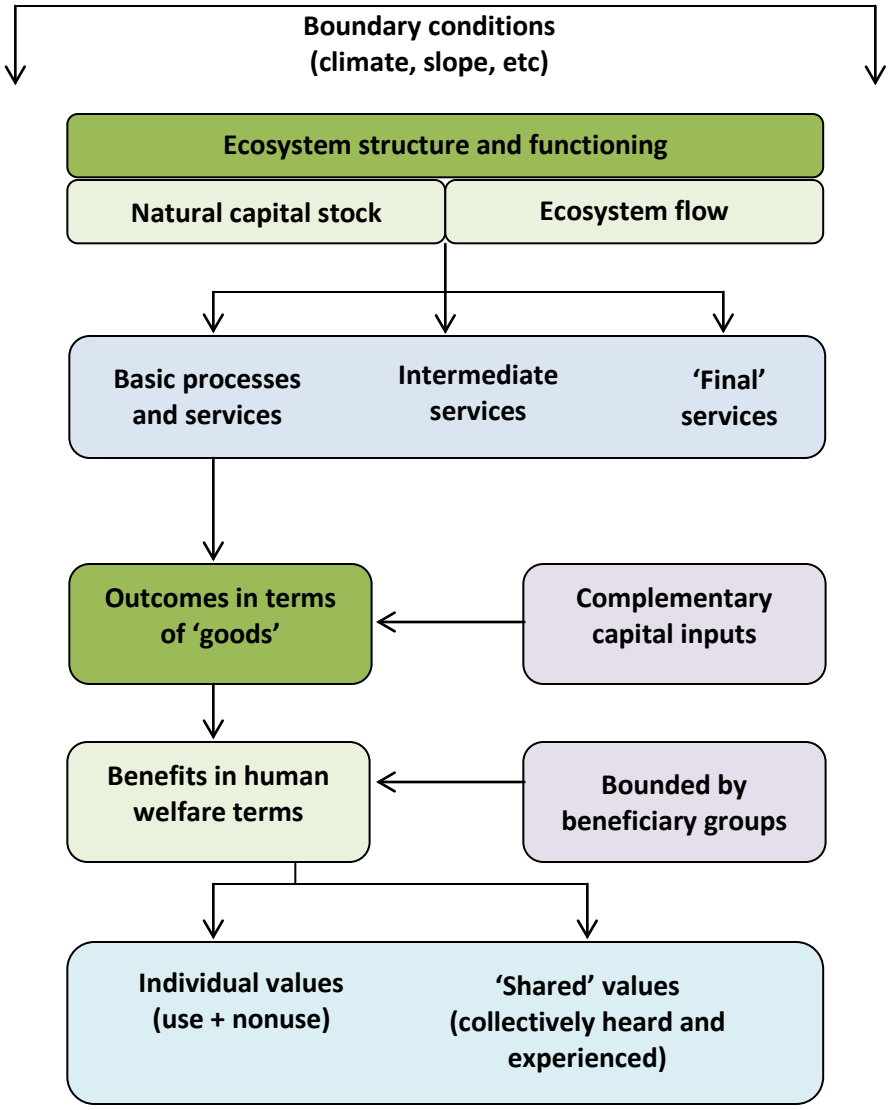

Figure 4.4. Ecosystem Services Conceptual Framework.

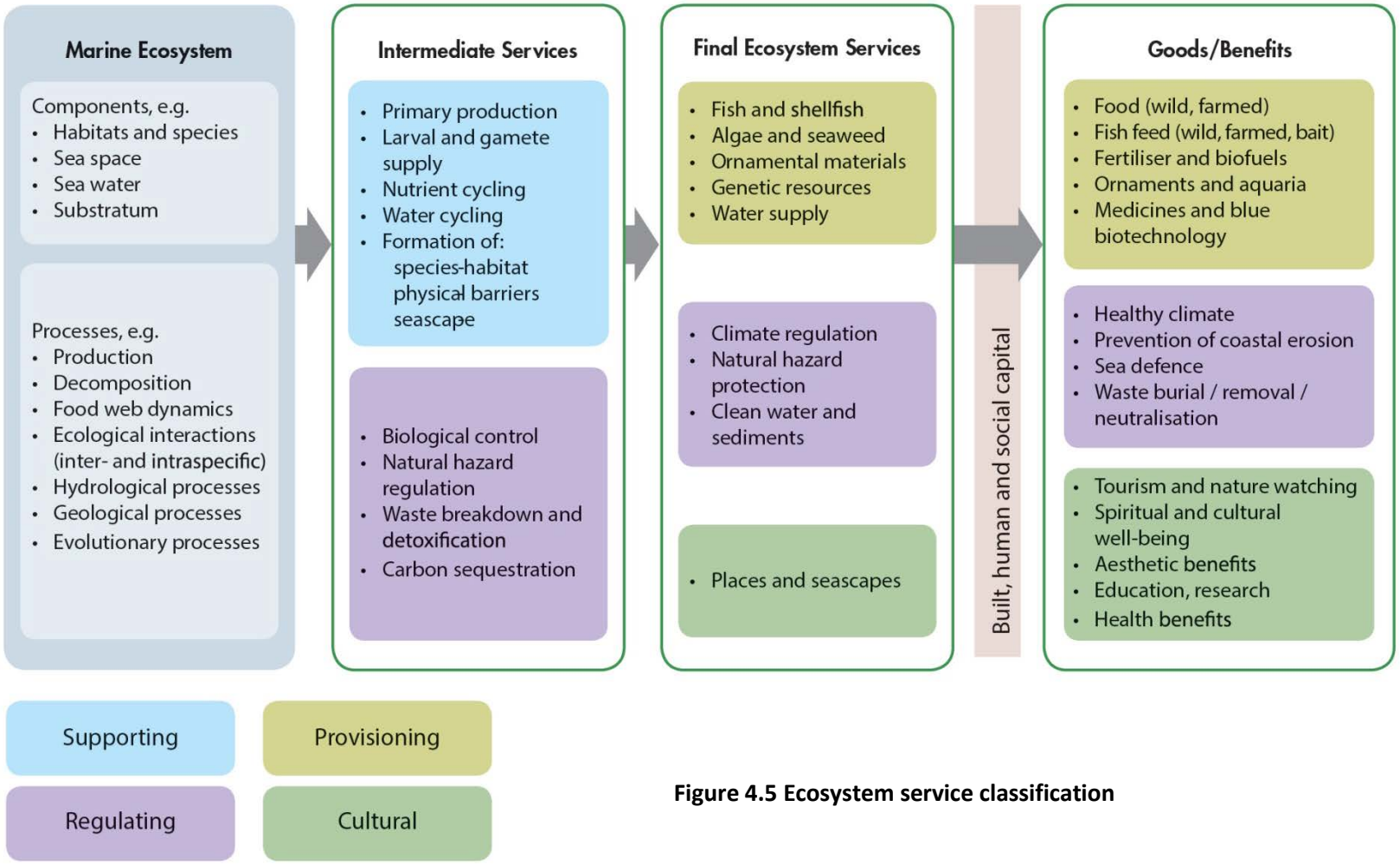

Figure 4.5 Ecosystem service classification

4.2.4 Coastal ecosystem processes and ecosystem services: goods/ benefits and values.

Ecosystems are dynamic systems made up of living and non-living components that interact with each other by way of complex exchanges of energy, nutrients and wastes. These exchanges are driven by the physical, chemical and biological processes or attributes that are characteristics of a particular ecosystem and its functioning. Ecosystem processes and functions include, for example, cycling processes such as nutrient cycling, nitrogen fixation and carbon sequestration which broadly map on to the 'intermediate services' concept in the classification system adopted here to facilitate monetary valuation: the basic ecosystem structure and processes combine to produce intermediate services and final services which can lead to goods (benefits) that are consumed by humans, or which are essential for human survival (MEA, 2005). Complementary assets (e.g. time, energy, finance or skills) also usually have to be combined with the natural capital to yield benefits. The intermediate services categorisation is used in order to clearly demarcate (in valuation terms) final services in order to avoid double counting. It is changes in the provision of services that we are interested in measuring and incorporating into economic and social analysis.

Depending on the precise definition used, coastal zones, for example, occupy around 20% of the earth's surface but host more than 45% of the global population and 75% of the world's largest urban agglomerations. The functioning of UK coastal and related marine areas is maintained through a diversity of ecosystems, e.g. salt marshes and other wetlands, sea grasses and sea weed beds, beaches and sand dunes, and estuaries and lagoons. This natural capital stock provides a range of services, such as nutrient and sediment storage, water flow regulation and quality control and storm and erosion buffering (see **Figure 4.5.**) (Crossland *et al.*, 2005).

Coastal zone ecosystems are impacted by dynamic environmental change that occurs both ways across the land-ocean boundary and their essential functioning depends on the connectivity with the catchment and the open ocean (Elliot & Whitfield, 2011). The natural and anthropogenic drivers of change (including climate change) cause impacts ranging from ocean acidification, coastal erosion, siltation, eutrophication and over-fishing, to expansion of the built environment and risk of inundation due to sea level rise. Globally, all coastal zone natural capital assets have suffered significant loss over the last three decades (e.g. 50% of fresh and salt water marshes lost or degraded, 35% of mangroves and 30% of reefs) (MEA, 2005). The consequences for services and economic benefits (value) of this loss at the margin is considerable, but has yet to be properly recognised and more precisely quantified and evaluated (Daily, 1997; Turner *et al.*, 2003; Barbier *et al.*, 2008; Mäler *et al.*, 2009).

Many definitions and classification schemes for ecosystem services exist (Costanza *et al.*, 1997; Daily, 1997; Boyd & Banzhaf, 2007). One of the most widely cited is the Millennium Ecosystem Assessment definition, which describes ecosystem services as 'the benefits that people obtain from ecosystems'. It classifies ecosystem services into: *supporting* services (e.g. nutrient cycling, soil formation, primary production), *regulating* services (e.g. climate regulation, flood regulation, water purification), *provisioning* services (e.g. food, fresh water), and *cultural* services (e.g. aesthetic, spiritual, recreational and other non-material benefits). This framework provides a platform for moving towards a more operational classification system which explicitly links changes in ecosystem services to changes in human welfare. By adapting and re-orienting this definition it can be better suited to the purpose at hand, with little loss of functionality. Wallace (2007), for example, has focused on land management,

while Boyd and Banzhaf (2007) and Mäler *et al.* (2009) take national income accounting as their policy context. For economic and social valuation purposes the definition proposed by Fisher *et al.* (2009) clarifies the distinction between ecosystem services and benefits: *ecosystem services are the aspects of ecosystems utilised (actively or passively) to produce human well-being*. Fisher *et al.* (2009) see ecosystem services as the link between ecosystems and things that humans benefit from, not the benefits themselves. Ecosystem services include ecosystem organisation or structure (the ecosystem classes) as well as ecosystem processes and functions (the way in which the ecosystem operates). The processes and functions become services only if there are humans that (directly or indirectly) benefit from them. In other words, ecosystem services are the ecological phenomena, and the good/benefit is the realisation of the direct impact on human welfare. The key feature of this definition is the separation of ecosystem processes and functions into intermediate and final services, with the latter yielding welfare benefits.

An intermediate service is one which influences human well-being indirectly, whereas a final service contributes directly. Classification is context dependent, for example, clean water supply is a final service to a person requiring drinking water, but it is an intermediate service to a recreational angler. Importantly, a final service is often but not always the same as a benefit. For example, recreation is a benefit to the recreational angler, but the final ecosystem service is the provision of the fish population. This approach seeks to provide a transparent method for identifying the aspects of ecosystem services which are of direct relevance to economic valuation, and critically, to avoid the problem of double-counting.

The policy context to which the analysis relates is also very important and influences the way in which the ecosystem classification can be utilised. To take an example, an estuary and coupled catchment characterised by, among other economic activities, intensive agricultural regimes. The estuary has extensive wetlands, salt marsh and mudflat areas which can provide a set of ecosystem services. Given the impacts of intensive agriculture, for example, heavy nutrient N and P runoff, the wetlands can provide valuable services such as nutrient cycling. If for example, national policy includes a provision to increase wetland habitat and the services it provides, in a cost benefit analysis (CBA) of this policy option the nutrient cycling service provided by the wetlands would be treated as an intermediate service contributing to the provision and value of final services e.g. better water quality. This cleaner water may then lead to enhanced recreation and amenity benefits, or improved fisheries productivity, which can be assigned a monetary value.

A change in the policy context, however, can change the way in which the ecosystem service classification is used. Assume the estuary is already subject to an official (national/international) water quality standard provision, which it is failing and the policy option under consideration is how best to meet the standard. Now cost effectiveness analysis (CEA) would be deployed to determine the least cost way of achieving the pre-existing water quality standard. In this context the nutrient cycling service provided by an increase in the wetlands via re-creation, would be focused on and compared with, for example, the cost of enhanced sewage treatment processes and facilities, or changes in agricultural regimes imposed on farmers (e.g. nitrogen zoning).

A pragmatic stance was taken within the UK NEA in order to bring the ecosystem services concept more fully into the collective consciousness of government (particularly finance ministries) and business. The methodology therefore deliberately allows for the monetary valuation of the outcomes from 'final' ecosystem services. This stance was pushed further, given the precautionary principle, in the sense that it was judged that sufficient scientific and socio-economic information exists to justify starting to

explicitly manage our ecosystems more sustainably and that there is a net benefit from such action. At the same time due recognition needs to be given to the danger of threshold effects because of the scientific uncertainty which shrouds how certain ecosystems may be adversely affected by human development pressures causing them to unexpectedly collapse or lose significant productivity potential.

The assessment and valuation of ecosystem stock and flow situations is therefore not a straightforward task. The monetary valuation of stocks and flows in particular is complex and has to rely on a range of accounting and socio-economic approaches, together with an underlying natural science understanding. Some services will not be amenable to monetary valuation, and the use of coastal resources and their conservation is often highly contested involving different interest groups. Coastal areas are also socio-cultural entities, with specific historical conditions and symbolic significance. The values expressed for such cultural entities may well manifest themselves through collective social networks such as groups, communities and even nations. They may not be best identified through an individual's monetary valuation, but through group deliberation and shared values in quantitative or qualitative terms, or through other evidence sources, e.g. archives. We take a closer look at 'shared values' in **Section 4.2.11.3.**

4.2.5 Decision Support System (DSS): Practice.

The DSS process needs to be composed of a number of sequential (depending on the exact policy/issues context) but overlapping components:

- a scoping exercise to establish baseline ecosystem and co-evolving socio-economic systems conditions and trends, together with a focused attempt to identify 'key' policy contexts/issues (**Sections 4.2.6 & 4.2.7**);
- a futures assessment through the use of scenarios covering prevailing conditions and alternative future states (**Sections4.2.8 & 4.3**);
- the selection and development of appropriate functionally related indicators of ecosystem state (the stock position) and changes in services (the flow position) supply over time (**Sections 4.2.9 & 4.4**);
- the deployment of 'tools'(including models) to enable a scientific, economic and social appraisal of policy options, including distributional concerns and the use of deliberative methods and techniques to foster social dialogue across interest groups (**Sections 4.2.10, 4.2.11, 4.5 and 4.6**);
- appropriate formatting of appraisal data, assumptions and findings (**Sections 4.2.12 and 4.7**); and
- setting up adequate monitoring and review procedures.

We look at the main components of the DSS below.

4.2.6 Scoping environmental change in coastal zones

The underlying activity-pressure-impact chain characteristic of coastal zones (Crossland *et al.*, 2005) can be been expanded to form the Drivers, Pressures, State changes, Impacts and Policy response (DPSIR) framework. Further, because of the continuing confusion between the S being State and State Change and the I being Impact (on the natural system) and Impact (on the human system) (Atkins *et al.*, 2011), the original formulation has been further modified to the DPSWR approach where W replaces I as impact on human welfare (Turner *et al.*, 1998; Cooper, 2013).

The DPSI(W) framework can help to scope in a standardised fashion policy and management contexts in order to get a better understanding of this environmental change process and what it means in

ecosystem service terms. This established scoping methodology can combine data about environmental change drivers and pressures with causal mechanisms which result in environmental state changes, and impacts associated with human welfare gains and losses. Feedback loops between policy responses and other components of the change process are also encompassed within the approach to avoid overly linear thinking as individual and societal innovation often occurs in a non- linear and in sometimes surprising ways. The approach first developed to classify and organise environmental indicators has proved to be a useful heuristic in wider environmental management contexts (Turner *et al.*, 1998). The scoping exercise has to be sufficiently robust to capture all the main drivers of change and behaviour incentives across multiple actors, jurisdictions and agencies. While it is the case that coastal and marine system issues can be complex and that a range or combination of variables influence human interest individuals and groups, under any given governance system, partial decomposition of problems is possible (Ostrom, 2007). However, the information provided by the DPSWR process will require further refinement to include a specific focus on ecosystem services and in order to highlight 'key' contexts and issues. The impacts/welfare stage needs to be specifically calibrated in terms of ecosystem services and interactions/feedbacks (Kelble *et al.*, 2013). **Section 4.2.7** illustrates some 'key' coastal zone management and trade-off situations.

Figure 4.6. illustrates the DPSIR framework in standard form, including feedback loops between Responses, and Drivers and Pressures, and recognition that there are natural pressures on ecosystems, which can lead to State Changes. Defining boundaries requires due care and attention, because pressures on the system can be locally, regionally or internationally managed pressures (power generation, fisheries, etc), or exogenic unmanaged pressures (climate change, volcanic eruptions, geomorphic isostatic readjustment, etc). The latter case, in contrast to the former, is one of bounded rationality (i.e. taking action with limited information on a 'learning-by-doing' basis) since their complexity is such that we do not yet have sufficient knowledge of how and why change occurs in such systems, and so our response is not of the management of the pressure but of the consequences of that pressure; in the case of endogenic managed pressures, we may be able to manage both the causes and the consequences (Atkins *et al.,* 2011).

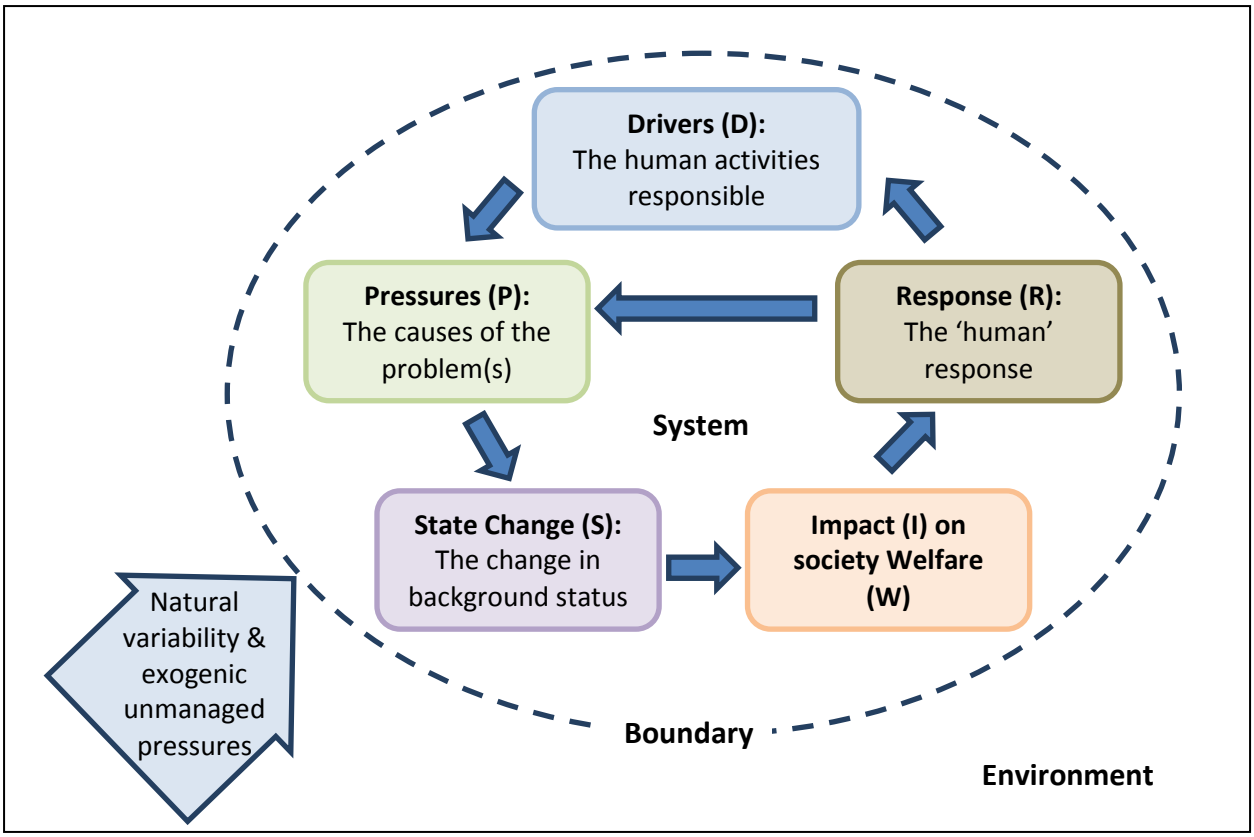

Figure 4.6. The DPSI(W)R framework. DPSI(W)R can be explicitly focused on ecosystem services through the S and I(W) stages.

The DPSIR framework has been widely used to assess and manage the impact of policy changes and associated problems; however, a change is evident in recent applications of the approach: an expert-driven, evidence focused mode of use is giving way to the use of the framework as a heuristic device to facilitate engagement, communication and understanding between different stakeholders (Cooper 2013; Kelble *et al.*, 2013). The application of scenario analysis to the DPSWR framework can be a useful way to further embed the DPSWR into the DSS for management. The state changes step in the framework can be further developed in termsof a natural capital asset check (see WP 1 Report).

4.2.7 Key policy issues

Identification of a relevant policy issue is a key stage of the management process. The rationale for government involvement in environmental management can be market failure, where existing markets create negative effects that are detrimental to a society at large, either now or into the future; and where government interventions can lead to socially more optimal outcomes.

The *framing* of a policy issue is necessary in order to enable identification of appropriate decision support processes and suitable policy instruments. Typical contemporary policy issues within the regional seas and coastal zones, and which are at the core of the need for better policy tools and governance regimes are diverse**.** For example, increase in human population size may lead to increase building activities in risk prone zones, including more artificial defence structures, which in turn can lead to the destruction of natural habitat or arable land. Increased shipping activities may lead to higher

pollutant discharges, antifouling paints or even oil spills in case of accidents, which directly affects sea biodiversity and coastal water quality with consequences on the fishery and tourism sectors. Aquaculture and wind farm development may lead to pollution and loss of habitat and biodiversity, which consequently affects goods and benefits such as fisheries and recreation, either directly or by providing a stepping stone for invasive species. Finally, the on-going (terrestrial) consumption of fossil fuels leads to climate change, which manifests itself in coastal and marine areas in sea temperature increase and sea level rise, and can have adverse effects on coastal safety, wild species diversity, human health (through toxic algae blooms), etc.

4.2.8 Coastal and marine futures scenarios

While future uncertainty will always remain problematic, scenario analysis (typically based on a 'business as usual' (BAU) baseline trend assessment, against which a range of different future paths can be assessed) offers a way of coping with uncertainty and provides policy relevant decision information on plausible future states of the world (**Figure 4.7.**). Section 4.3 discusses the scenarios for coastal and marine habitats developed in Work Package 4 in more detail.

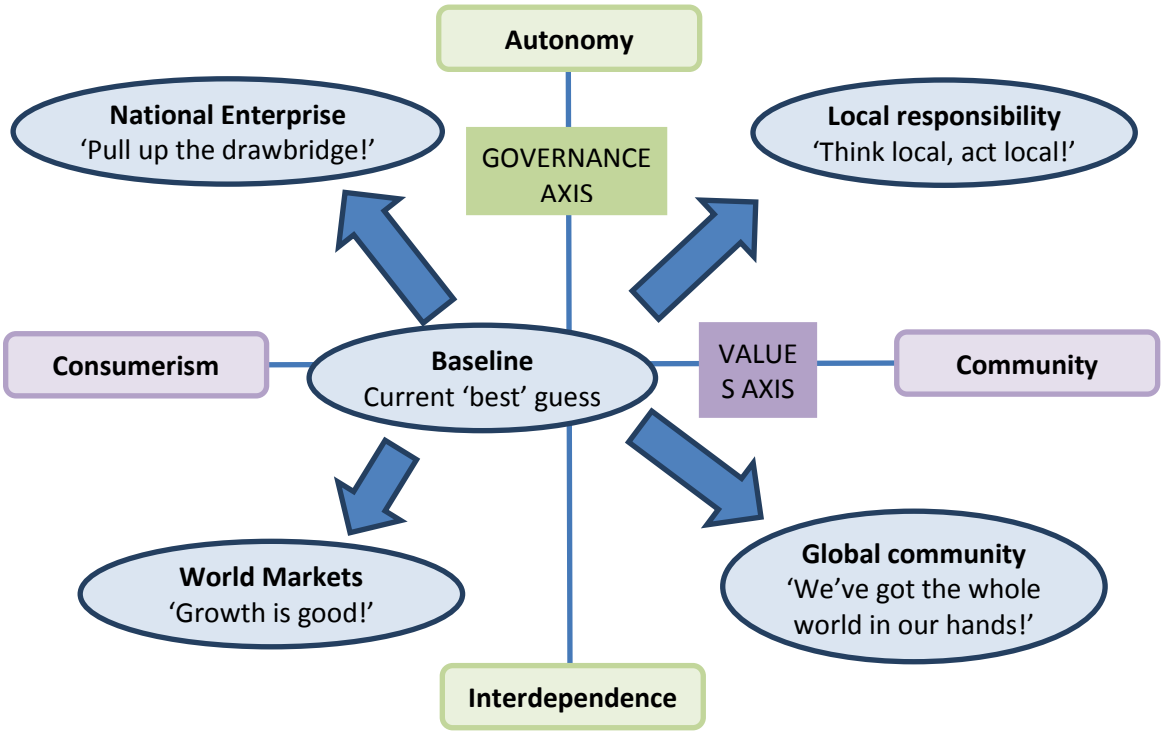

Figure 4.7. Scenarios adopted for the EU Project 'European Lifestyles and Marine Ecosystems' (ELME). These are based on the SRES scenarios employed for climate change studies and describe plausible alternative worlds. These worlds lie within two axes describing a spectrum of human values and attitudes towards governance. The 'Baseline' scenario is the 'best guess' of Business as Usual (BAU). Source: Langmead *et al.* (2007).

4.2.9 Indicators

The future challenge in the EU is the joint implementation of the WFD and MSFD with the former focusing on the protection of the system according to chemical status and five biological quality elements (four in the coastal zone), whereas the MSFD focuses on 11 descriptors, each of which can be linked to show a hierarchy (see Borja *et al.*, 2010). The WFD is regarded as a 'deconstructing structural' approach, whereby the indicators are more easily related to the structural ecosystem components, whereas the MSFD apparently will relate to functioning of the system and a more well-defined set of pressures along the activity-pressure-impact chain (Borja *et al.*, 2010).

The MSFD has stimulated new work into appropriate indicators linked to the eleven descriptors of the environmental change process as it affects coastal and marine ecosystems (stock and flow) and their services provision. Functional indicators are required, for example, across media, spatial location, hydrological function and biological function. **Section 4.4** presents the indicators developed in Work Package 4 for the assessment of coastal and marine ecosystems.

4.2.10Models

An important component of the adaptive approach and DSS is the development of models. A number of different types of models can be deployed, ranging from formal scientific models of land use change in catchments with links via nutrients and other factors into models for estuaries and coastal waters, to conceptual models which are simple ways of highlighting and eliciting human perceptions about how a system functions. The latter allow a dialogue between experts, stakeholders and the public which conveys information, identifies 'contested' issues and provides the opportunity to reinforce or modify perceptions and expressed values (Turner, 1999). Underpinning the approach is a requirement to collect empirical data and metadata on ecosystem functioning and service provision, together with an understanding of the distribution of ecosystem benefits (who gains or losses in any environmental change situation) and governance contexts. We review the available models for coastal/marine systems in **Section 4.5**.

4.2.11Economic and Social Appraisal

The application of economic and social appraisal of projects, policies, programmes or courses of action in the coastal context can only take place after policy issues have been identified and highlighted within given spatial and temporal scales, and scenarios and evaluative criteria have been established and legitimised within the dialogue process. Once agreed, the policy issues and scenarios chosen then provide the backdrop and framework within economic and social appraisal can take place. However, this is not a one-way process. Ideally, feedback should occur between all stages of the assessment process and the deliberative procedures set up with stakeholders, since concerns that are thrown up by the dialogue can help to refine the policy issues, leading to acceptable interventions and scenarios that resonate with most stakeholders and interest groups.

4.2.11.1 Environmental impacts, welfare and economic values

Once policy issues and scenarios are established, the next stage of the process is to determine all the relevant impacts that will take place under the scenarios considered. These impacts relate to changes in the provision of ecosystem final services and goods (which could include, for example, the carbon

storage functions of coastal mudflats) and other, more conventional goods (such as commercial fish catch or shellfish harvested from coastal mudflats). Primarily, economic assessments are concerned with those impacts on goods that can be valued in monetary terms. However, this does not mean that all impacts can be incorporated into such an analysis – it may not be possible to value all impacts in this way, because of practical or ethical considerations. Hence we consider that economic assessment provides just one strand of an overall integrated (sustainability) analysis, with other strands being supplied by assessments from social/ deliberative and ecological perspectives (such as multi-criteria analysis (MCA), participatory GIS, deliberative fora, deliberative monetary valuation, see WP 5 Report and WP 6 Report). It is also the case that the sustainable provision of the flow of final services and related goods and benefits depends on the maintenance of an ecosystem processes with adequate carrying capacity and resistance and resilience characteristics.

The core of the economic assessment process is to determine how changes in ecosystem services provision are translated into changes in welfare (which can be positive or negative, i.e. benefits or costs). This is achieved by placing a monetary value on those changes and aggregating these values together to arrive at an overall change in value for the environmental and policy scenarios considered.

In the economic literature, a number of issues can be identified as key to the *appropriate* economic valuation of ecosystem services. These are: spatial and policy context explicitness, marginality, the double-counting trap, non-linearities in benefits, and threshold effects (see **Figure 4.8.**).

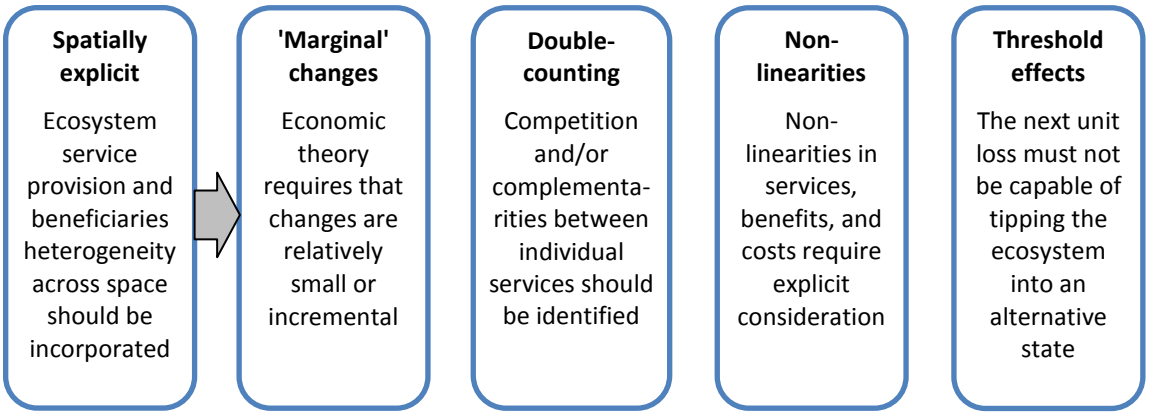

Figure 4.8. Ecosystem Services Sequential Steps: A framework for appropriate economic valuation. Source: Morse-Jones *et al.* (2008).

Therefore to be most useful for policy, services must be assessed within their appropriate spatial and policy context and economic valuation should provide marginal estimates of value (avoiding double counting) that can feed into decisions at the appropriate scale, and which recognise possible non-linearities and are well within the bounds of safe minimum standards (MEA, 2005; Turner *et al.*, 2003).

Ecologists use the term *value* to mean 'that which is desirable or worthy of esteem for its own sake; something or some quality having intrinsic worth'. Economists use the same term to describe 'a fair or proper equivalent in money, commodities, etc', where *equivalent in money* represents that sum of money that would have an equivalent effect on the welfare or utilities of individuals. A number of ecosystem goods can be valued in economic terms, while others cannot because of uncertainty and complexity conditions. The notion of total economic value (TEV) provides an all-encompassing measure of the *economic value* of any environmental asset. It is important to note however that TEV is always

less than total systems value (TSV). A minimum configuration of ecosystem structure and process is required before final services and goods can be provided. We take a closer look at the TEV concept and related issues in **Appendix 4.2**.

It is important to note that the value of nature concept is usually interpreted in economic analysis in terms of individuals and their preferences and motivations. The value concept can also however be viewed in a collectivist way, and expressed or elicited in a collective way (i.e. shared values, see WP 6 Report). Following WP 6 Report, cultural or societal values, as well as communal and group values, include principles and values as well as a shared sense of what is worthwhile held by members of a society, community or group. This is in terms of motivations and preferences assigned to groups and culturally transmitted and assimilated over time as social norms. These shared values may not be capable of meaningful monetary expression, but nevertheless they significantly signal that human well-being and quality of life is a function of both individual wants satisfaction and the meeting of a variety of social, health related and cultural collective needs. Cultural values include shared values fostered by and within 'groups' often acquired over long periods of time and often connected to specific local places, e.g. East-Anglian landscapes with traditional windmills.

4.2.11.2 Stock versus flow values

The distinction between ecosystem services stocks and flows has also to be reflected in the economic valuation approach adopted. The paper in the journal Nature by Costanza *et al.* (1997) estimated the value of global ecosystem services at $33 trillion and led to a protracted debate and controversy over the 'true' value of the natural environment. The value of nature is a multidimensional concept which includes monetary value but also more qualitative measures. The complete 'commodification' of nature is an ever present danger to be avoided according to critics of monetary valuation. The position adopted here is that many (but not all) ecosystem services can be meaningfully expressed in monetary terms and that this type of calculus has 'political' purchase which can be used to further conservation efforts in the real world.

The Costanza *et al.* (1997) global ecosystem services estimation has been attacked on a number of grounds including that the aggregate value was not necessarily the sum of the parts, and that US$33 trillion was more than global income and therefore peoples' ability to pay (Heal, 2000). Further work (Howarth & Farber, 2002) sought to defend the Costanza *et al.* approach by arguing that the estimates of ecosystem services value were analogous to National Income Accounting entities such as GDP with a constant set of value weights. The underlying rationale here is that the aggregate measure is a quantity parameter (the stock concept), and, while it is related to value, it does not directly value the planet's ecosystem services in total. In this sense it is an accounting price measure of the quantity of ecosystem services holding prices constant, where the measures are not based on economic theory but on accounting rules. In this stock accounting context the criticism related to peoples' budget constraint and ability to pay is not relevant, because the measure is based on virtual (not real) prices and virtual incomes (i.e. incomes adjusted to enable individuals to hypothetically pay for the services).

For the income and expenditure accounts to balance, the total expenditure must be less than actual and virtual income. The current extent of European coastal blue carbon (the carbon storage service provided by salt marshes and sea grasses) has, for example, an accounting stock price (value) of about US$180 million, valued against a Social Cost of Carbon estimate (Luisetti *et al.*, 2013a). Such total (stock) values can be estimated and compared for two different points in time as a heuristic to help to appreciate the

change in natural capital. This viewpoint is, however, controversial and is not supported by many mainstream economists. For them the only relevant measure is the marginal economic value.

For economic valuation it is important to be able to quantify and evaluate gains or losses in stock assets and consequent service flows (analogous to net GDP).[1] Now instead of holding prices constant, we seek to determine marginal economic value as it relates to an incremental increase in a set of ecosystem services over time and space. When the ecosystem final services value relates only to non-market services, it can be combined with GDP (in the same way as relevant pollution and other externalities are internalised) to yield a more green GDP measure. The present value of a discounted flow of ecosystem services values can contribute to stock of wealth accounts such as the Inclusive Wealth account (UNU-HDP & UNEP, 2012). An important consideration is that the flow and stock values as explained in the above serve different purposes, and they are not comparable and should not be added up.

The studies reviewed in this report all provide estimates of the value per year, i.e. flow values. But a separate and complementary ecosystem services account/index may also be a worthwhile objective. Overall, the future goal should be to measure and value both service flows and to predict changes in stocks (ecosystem health) which condition future flows.

4.2.11.3 Shared values

Valuing the contribution that ecosystem services make to human well-being cannot be reduced to individual preferences (WTP) and motivations alone. Ecosystem services may also have collective meaning and significance. So-called 'shared values' concern values humans hold for ecosystem services as 'citizens', i.e. part of a collective entity governed by social rights and wrongs. Shared values can be provided by groups, communities and societies as a whole and may be considered as shared principles and virtues (see WP 6 Report). They may differ in intention from purely self-regarding interest to include other-regarding concerns and therefore encompass a consideration of the ethical arrangements which guide society's concern for nature, place, landscape and seascape, and include motivations such as altruism, bequest value and existence value (Fish *et al.*, 2011). Some analysts would also see aesthetic considerations as an additional value dimension. Society's acceptance of the reliability and legitimacy of decision making processes that have been informed by technical DSSs and have highlighted trade off dilemmas can in certain contexts be heavily influenced by whether shared values have or have not been explicitly recognised and accounted for in the political process.

Shared values often have to be elicited through an interpretative approach which relies on qualitative expressions of value e.g. through the interpretation of documents and media, but also via group discussion, learning and deliberation. Key techniques are deliberative (non-)monetary valuation and participatory MCA, which hold much promise in terms of a systematic and integrated treatment of utilitarian and other ethical positions, as well as aesthetic considerations. Systematic large scale surveys (e.g. Potts *et al.,* 2011) can begin to unwind broad social values and inform further analysis. They remain, however, at an experimental stage of evolution. It is important to note that while techniques are evolving to better understand shared values, the social learning mechanisms are 'processes to be engaged in' facilitating policy deliberation among equal partners.

[1] GDP reflects the financial (market) value of all final goods and services produced within a country within a certain period. Net Domestic Product (net GDP) is GDP net of the depreciation on capital goods, and thereby reflects how much capital has been consumed over the year.

4.2.11.4 Discounting, equity and distributional considerations

It is often necessary to choose between options that differ in temporal patterns of costs and benefits, or that differ in their duration. Discounting provides a common matrix that enables comparison of costs and benefits that occur at different points in time. Use of discounting yields an outcome in which future costs and benefits are valued less highly than those that occur in the present, and the procedure is integral to CBA and CEA. The choice of the discount rate can have a significant effect on the economic viability of management options and their relative economic ranking. It signals the rate at which future consumption is to be traded against consumption in the present. Use of a 'high' positive rate of discount discriminates against the future and in project terms against options that involve high initial costs and a stream of benefits that extends far out into the future (e.g. coastal wetland creation, restoration, or maintenance within a coastal defence or protection strategy). Instead it tends to favour projects that have immediate benefits and delayed cost burdens (Turner, 2007). But while a low discount rate favours the future, this may be politically and morally questionable if immediate wellbeing increases are slowed or compromised altogether and the burden falls disproportionately on the poor.

The discounting question raises a number of much deeper ethical and strategic considerations related to equity and fairness principles and practice. Fairness in contemporary society (intra-generational equity) is sidestepped in conventional applications of CBA via the acceptance of the economic efficiency criterion which weights all benefits and costs equally, regardless of whether they affect rich or poor in society (known as Potential Pareto Improvement as determined by the Hicks-Kaldor compensation test) (Gowdy, 2004; Turner, 2007). We make a case for actual compensation (financial and environmental), especially given the 'contested' nature of environmental change in coastal zones, in a later section (see **Sections 4.2.11.6 and 4.2.12**). The debate around discounting has a long history and involves some difficult ethical questions, we summarise some of this in **Appendix 4.1**.

4.2.11.5 Efficiency and other decision criteria

Most methods of economic assessment are concerned with determining the efficiency of policy options, where efficiency is defined in an economic sense in which the most efficient solution is the one that increases overall welfare to the greatest extent. Efficiency is not necessarily associated with equity (i.e. questions of where welfare benefits or costs fall; e.g. on particular sectors of industry, certain social classes, certain geographical areas, etc). However, sustainable solutions must consider both equity and efficiency. Given the 'contested' nature of coastal socio-ecological resource systems (Ostrom, 2007), questions of trade-offs, social justice, equity and compensation are likely to loom large in public debate.

Appropriate DSSs can therefore be informed, for example, by a better understanding of relevant social and policy networks (Bodin & Crona, 2009; Borgatti *et al.*, 2009; Bainbridge *et al.*, 2011); and also via methods and techniques encompassing multiple values and decision criteria. Economic assessment methodologies can be modified to incorporate equity issues (e.g. via the application of weights to costs and benefits), and the economic analysis itself can be augmented by a wider trade-off analysis, for example using MCA or deliberative (non-)monetary valuation techniques.

DSSs and their component methods and techniques such as CBA, CEA and MCA, require the acceptance of different assumptions about the capacities and motivations of the individuals involved, and the role the methods play in framing/scoping the assessment process. From an institutional perspective, CBA and other methods can be characterised as value articulating institutions, in the sense of rule structures facilitating value (Vatn, 2009). If the existence of plural rationalities is accepted, the role of such

institutions is to signal which rationality is expected. The choice of assessment approach is , for example, related to whether the benefits and costs involved are linked to an exchangeable commodity or some kind of public good; if it is the allocation of a public type good that is being contested then deliberative methods may play a useful role, although they need to be tailored to suit the situation e.g. citizens or stakeholder forum or a hybrid. **Section 4.2.12** sets out a different appraisal format, the 'balance sheet' approach given a pluralistic perspective and the increasing number of 'wicked' policy contexts that challenge decision makers in coastal zones.

Early on in environmental evaluation it is important to differentiate between seeking to determine the most efficient plan, project, policy or a programme of coastal and marine intervention, or a more constrained CEA. In the latter context, a range of options are usually assessed to see which yields the desired (determined *a priori*) outcome, e.g. achievement of an official water quality standard, at least cost to society. The main distinction between CBA and CEA is that the desired outcome(s) is determined *a priori* in CEA but not in CBA. However, for industry and the regulators, these also have to be placed within a context of Best Practicable Environmental Option (BPEO). BPEO has been defined by the Royal Commission on Environmental Pollution (1988) as "the outcome of a systematic consultative and decision making procedure which emphasises the protection and conservation of the environment across land, air and water. The BPEO procedure establishes for a given set of objectives, the option that provides the most benefits or the least damage to the environment, as a whole, at acceptable cost, in the long term as well as in the short term". We outline in **Section 4.2.12** a particular sequencing of policy tools (methods/techniques) labelled the **'balance sheets' approach** which starts with CBA/CEA but then encompasses other complementary 'tools' to apply AM principles.

4.2.11.6 Policy response interventions

The policy response **interventions** (see also WP 8 Report) usually fall into a number of categories:

- **Mitigation of pollution and resource overexploitation problems** – the ecosystem service benefits that need to be valued are related to damage reduction and/or restoration measures, e.g. reduced flooding damage or sedimentation in navigation channels or restoration of wetlands, water treatment investment, changing farming practices in the catchments, etc.
- **Compensation for losers measures** - these may be financial as in the case of coastal erosion problems in England and Wales with, for example, the Pathfinder experimental scheme in which local authorities offered to pay 40-50% of the theoretical value of properties threatened by coastal erosion, based on the value of similar properties inland; or environmental compensation under a precautionary principle, safe minimum standards approach which can include project management on a portfolio basis (Barbier *et al.*, 1990) with so-called 'shadow' or 'compensating' projects; or habitat equivalency compensation measures (Roach & Wade, 2006).
- **Enhancement of marine/coastal zone ecosystem services** – actions which provide an increased provision of benefits, e.g. adaptation to change (see **Figure 4.9.**), which increases the output of some good such as creation of artificial reefs to provide erosion protection, or fisheries habitat and nursery which enhance productivity of the stock, or the reduction of conflicts among or between various users of coastal ecosystems via pricing schemes or zoning.
- **Preservation of unique marine/coastal ecosystems** – the benefits stem from setting aside and managing particular areas via Marine Protected Areas (MPAs) in order to preserve the natural ecosystem can be twofold. Use benefits e.g. visits to a nature reserve to observe nature or take photographs etc; and non-use benefits which are not related to visits but encompass option or existence values. The non-use values here relate to motivations which seek to conserve ecosystems

for future use (insurance value) and the continued presence of species and habitats from which people derive passive welfare. Shared values will also be important in this category.

- **Joint usage benefits -** within this last category of interventions, marine spatial planning and zoning have recently come to the fore, including the search for joint usage benefits. The UK Marine Policy Statement contains the following statement: "The Marine Plan should identify areas of constraint and locations where a range of activities may be accommodated. This will reduce real and potential conflict, maximise compatibility between marine activities and encourage co-existence of multiple users".

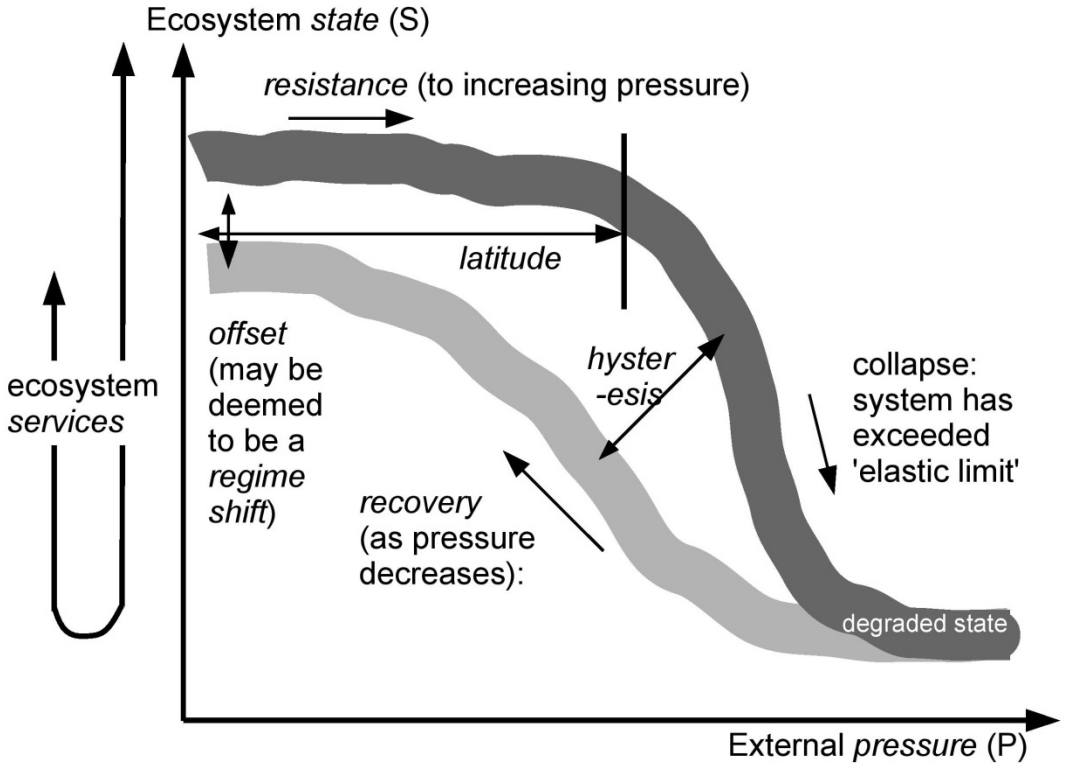

Figure 4.9. A conceptual model of changes to the state of a system with increasing pressure.
Source: combines ideas in pressure-state diagrams by Tett *et al.* (2007) and Elliot *et al.* (2007).

Ecosystem *adaption* to pressure is a complex process. It can occur at the population and species level as well as within trophic networks. Mechanisms are rarely well known in the case of marine ecosystems, and discussion is often conducted in terms of an emergent property, that of system *resilience*. This refers to the extent that the system maintains its integrity as external pressures increase (*resistance*), or regains that integrity when pressures relax (*recovery*). In this diagram the provision of services is shown as a function of ecosystem *state* (indicating integrity or health: see Tett *et al.*, 2013). Recovery, however, may involve change in ecosystem condition (sometimes called regime shift), so that restored services are not identical with those before system collapse.

There is a need to better understand the barriers to the achievement of joint net benefits, i.e. co-location situations in which multiple users or activities share the same impacts footprint (MMO, 2013). The decision to locate any given economic activity in a particular marine space will be conditioned by a

range of factors. At the core of this process will be an assessment of financial profit or loss potentially available to the economic agent (individual or firm) involved. However, the decision will be further constrained by existing and possible future legislation and regulation and wider social and environmental issues, such as, for example, loss of local employment or cultural identity when fishing activities are curtailed or lost; and environmental impacts including use and non-use loss if biodiversity is reduced. So the impacts (footprint) of co-location can be multidimensional and any assessment method must be able to accommodate this diversity if it is to offer improvement in and 'future proof' the DSS. The balance sheets approach framework set out below seeks to meet this need.

Two economic concepts, externalities and joint production, can be used in order to formally distinguish between the different possible categories of co-location. The 'technological externalities' concept refers to the indirect effect of an economic agent's consumption or production activity on the products, consumption or welfare of a different economic agent, and where the effect does not work through the price system. Externality effects can be positive or negative and quite diverse, including forms of pollution or contamination and interaction between different production activities. In the latter context, so-called 'joint production' cases can be identified. So multiple products may be produced under separate production processes, or several outputs may be produced from a single production process.

Three distinct categories of co-location for a given marine space can be identified using the economic concepts of externalities and joint production:

- No co-location – situations in which there are no feasible joint production possibilities and candidate activities generate negative externality effects; e.g. offshore wind farms and demersal fishing with beam trawls;
- Horizontal co-location – joint production possibilities exist and the candidate activities do not generate significant negative externality effects e.g. offshore wind farms and open water aquaculture; and
- Vertical co-location – no joint production possibilities and no negative externality effects, e.g. recreational fishing or boating in a MPA but limited to certain times of the year to protect fish spawning or biodiversity.

Finally, we turn to the question of how appraisal might be sequenced and how information can best be presented to policymakers.

4.2.12 Balance sheets appraisal format

If CBA or related methods are to continue to play a role in the policy process, then a more explicit focus on distributional issues (i.e. who gains and who loses from environmental change and consequent policy responses) is required. A two stage approach needs to be adopted in which the spread of costs and benefits across different affected individuals and groups in society needs to be accounted for and a weighting procedure applied. Project appraisals funded by economic development agencies have routinely included distributional weights but this practice has not been common place in other public sector applications. As a minimum, the way in which the CBA 'accounts' are set out and formatted needs to be changed in order to incorporate and highlight financial transfers and the distributional impact of costs and benefits across stakeholders. Krutilla (2005) has set out a tableau format which disaggregates the benefits and costs of a project or policy among stakeholders and records all inter-stakeholder financial transfers. It also serves to illuminate key issues such as the level of aggregation adopted and the project/policy accounting boundary.

Changing the accounts format is a necessary first step, but Kristrom (2005) has gone further and put forward a 'hierarchy of options approach' in which explicit distributional weighting is applied, based on a rule that requires higher weights on all costs and benefits accruing to socially disadvantaged or below average income groups. Alternatively, explicit distributional weights can be introduced to reflect the degree of inequality aversion present in society, by examining past public policy decisions, or the prevailing marginal rates of income tax (Atkinson *et al.*, 2000). We look at a particular way of formatting appraisal data and findings, the 'balance sheet' approach in **Section 4.7.**

The next section presents and discusses results from an expert workshop that aimed to assess the impact of several scenarios on the supply of ecosystem services by coastal and marine habitats into the future.

4.3 Coastal and marine futures scenarios

4.3.1 Introduction

This section discusses the scenarios of the supply of ecosystem services by coastal and marine habitats at 2060. In **Section 4.5**, we will look at models as tools for assessing and managing ecosystem services as they are impacted by climate change. Changes in UK society may influence such impacts, and the social changes might themselves be modelled. We first explore that possibility before presenting the expert assessment method that was used instead.

4.3.1.1 The nature of models

Conceptual modelling, the making of maps showing relationships amongst components of a system, seems to be a straightforward and widely comprehensible formalisation of an innate human ability. In contrast, the construction and use of numerical simulation models is a technically demanding activity grew out of abstract mathematical developments (such as calculus) and the idea of algorithms, and the invention and evolution of electronic computers and languages used to program them.

To 'run a model' or make a simulation is, often, to use numerical methods to solve sets of differential equations in which system state variables are expressed as functions of time. The model is a set of equations that is designed to refer to a particular spatial or conceptual domain; at the bounds of the domain are the boundary conditions that influence what happens inside the modelled system. The equation set may be very detailed, representing many processes and locations, as in the case of a food web model of a spatially heterogenous ecosystem. Alternatively, it may be a simple 'idealisation' of bulk processes or 'emergent properties' of the system under consideration. The following are needed to make a simulation: 1) the model equations expressed as algorithms and programmed into a valid set of computer instructions; 2) values for the parameters (the temporary constants) that are part of these equations; 3) a set of initial values of the system state variables; 4) time-series of boundary conditions, sometimes described as 'inputs', or 'forcing' (see **Box 4.A2.** for further explanation).

4.3.1.2 What is a scenario?

The meaning of 'scenario' is fuzzy. The word appeared in English in 1878, signifying a sketch of the plot of a play.[2] Only in 1962 did it gain the relevant meaning of an imagined situation, in relation initially to its military use for strategic planning.[3] The use of scenarios in planning was taken up in the 1970s by commercial organisations, notably Royal Dutch Shell, which continues to argue that *'the future is neither completely predictable nor completely random. Any meaningful exploration of possible future landscapes inevitably highlights alternative features or patterns. For over four decades now, Shell has developed and applied contrasting scenarios to help us consider the future more extensively and deepen our strategic thinking.*' (Shell Scenario Team, 2013) *'Scenarios are stories that consider "what if?" questions. Whereas forecasts focus on probabilities, scenarios consider a range of plausible futures and how these could emerge from the realities of today*.'[4]

[2] www.etymonline.com
[3] Wikipedia entry on 'Scenario planning'
[4] www.shell.com/global/future-energy/scenarios.html

In 2003, the MEA proposed to *'use scenarios to summarise and communicate the diverse trajectories that the world's ecosystems may take in future decades. Scenarios are plausible alternative futures, each an example of what might happen under particular assumptions. They can be used as a systematic method for thinking creatively about complex, uncertain futures. In this way, they help us understand the upcoming choices that need to be made and highlight developments in the present*' (MEA, 2003). Earlier, in 1994, the Intergovernmental Panel on Climate Change (IPCC) began to recommend the use of scenarios for the assessment of climate change impact. Following this, the UK Climate (Change) Impacts Project (UKCIP) argued in 2000 that '*different social and economic structures will affect sensitivity to climate change, as they affect the potential for response and adaptation. The impacts of future climates will also be fundamentally determined by future technology and governance structures*' (UKCIP, 2000)**.** One way to explore this would be to construct a set of internally coherent but different socio-economic scenarios, and to work out how climate change impact might be magnified or modified under each of these. In the UK NEA 2011, Haines-Young *et al.* (2011) proposed six scenarios (**Table 4.1.**) for this purpose, most of which correspond to the scenarios used in the present study.

4.3.1.3 Scenarios and models

In principle, scenarios might be used with models of the relationship between society and ecosystem services. **Figure 4.10.** is a conceptual diagram of the main components of such a model, framed in terms of DPSIR. It includes two types of scenarios: those for human emissions of 'Long-Lived Greenhouse Gases' (LLGHG) that contribute to the atmospheric greenhouse effect and thus to global warming; and those, called socio-economic scenarios, which refer to different possibilities for the organisation of societies on national and global scales.

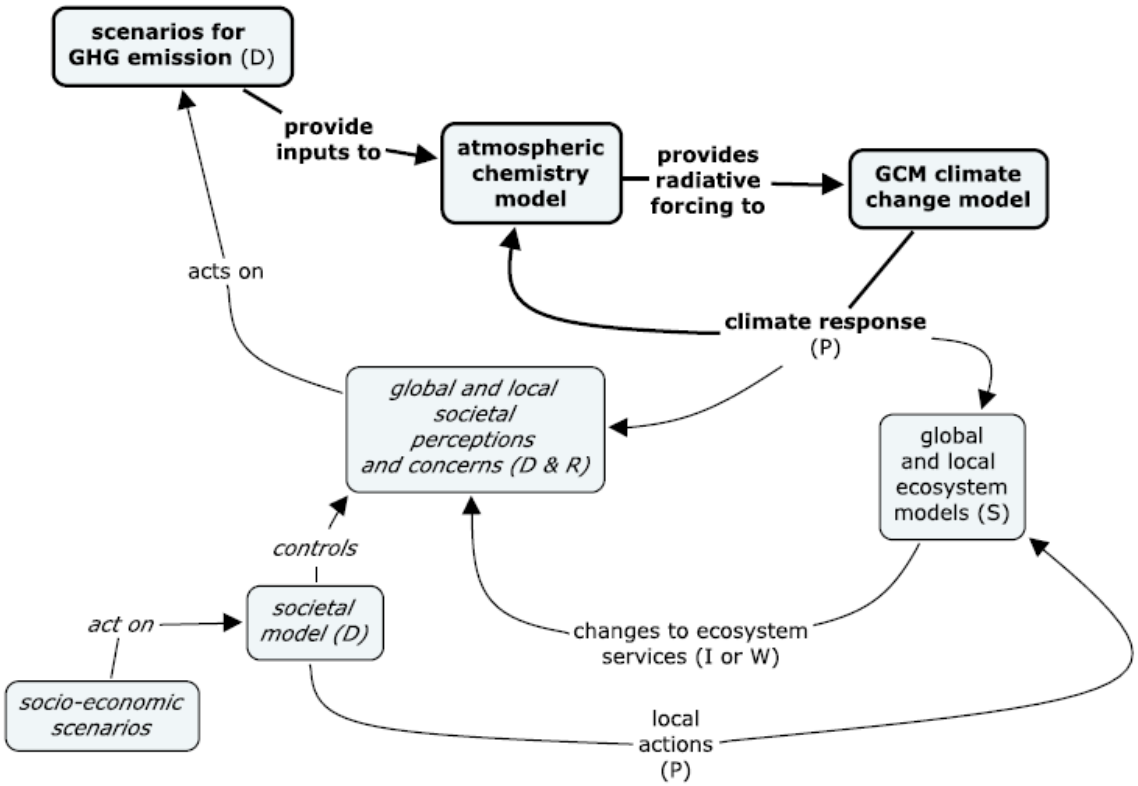

Figure 4.10. Conceptual model of a social-ecological system (SES). The model shows how two sets of scenarios influence climate change, ecosystem services, and societal impact on ecosystems. Each model component is annotated with a letter from the DPSIR framework: i.e. Driver (in society), Pressure on the State of ecosystems, Impact on society (or a change in Welfare of humans) via services, and consequently a Response in society.

The emission scenarios have been used, by way of models of atmospheric (and sometimes oceanic) chemistry, to predict changes in planetary and regional radiative forcing that feed through into climate change (**Figure 4.11.**). Each scenario has two aspects: first, a vision of change in global human society; and, second, a schedule of the consequent emissions of LLGHG. The vision is perforce fuzzy; the schedule is a concrete set of numbers for use as model forcing, or, to use a term defined above, as time-series of boundary conditions. **Box 4.1.** describes a widely used 'medium emissions' scenario.

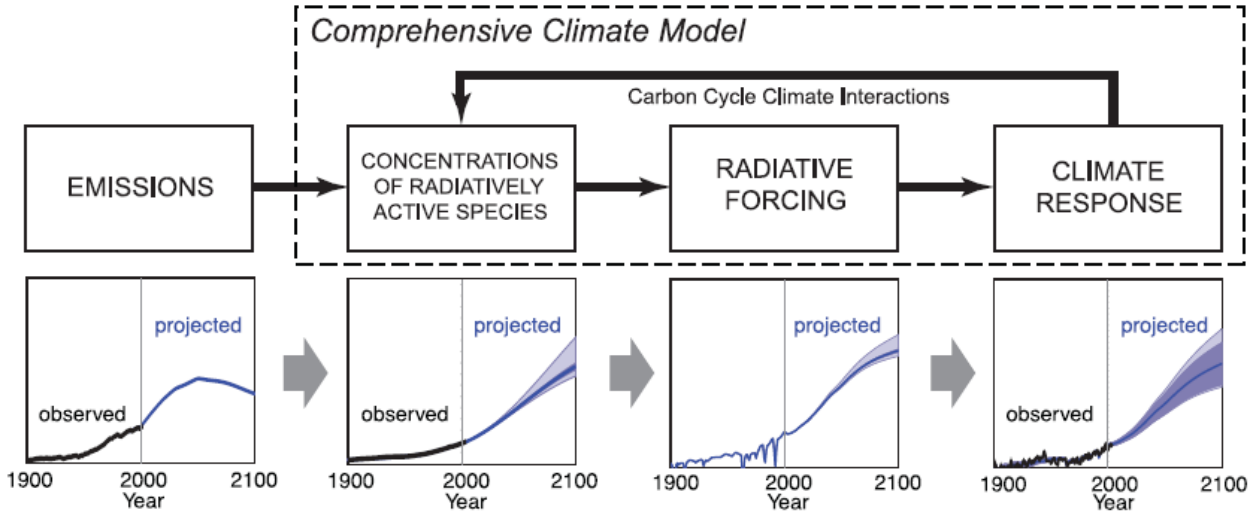

Figure 4.11. Steps from emissions to climate response contributing to uncertainty. An emissions scenario (such as A1B shown here) is a schedule of gases added to the atmosphere; the uncertainty in the predictions of climate change is the result of insufficient knowledge of relevant processes. The envelopes include results from simulations with multiple models that parameterise these processes in slightly different ways, and from groups of simulations with the same model but using a range of values of the key parameters. Source: Meehl *et al.* (2007), figure 4.10.1.

Box 4.1. The A1B emissions scenario[5]

The Intergovernmental Panel on Climate Change (IPCC), set up in 1988, published a Special Report on Emissions Scenarios (SRES) in 2000. The report describes four families of scenarios for human socio-economic development. The A scenarios assume market-oriented societies driven by desire for economic growth, while the B scenarios assume a greater influence by environmentalism. A1 and B1 assume moderate increases in global population, whereas A2 and B2 assume larger increases. The storyline for A1 '*describes a future world of very rapid economic growth, and a population that increases from 5.3billion in 1990 to peak in 2050 at 8.7billion and then declines to 7.1billion in 2100. Rapid introduction of new and efficient technologies is assumed, as is convergence among regions ...*' (Warren, 2009). Within this family, A1F assumes that energy continues to be generated mainly by fossil fuels, whereas A1T assumes a major shift to non-fossil-fuel sources. A1B is intermediate between these two. The SRES estimates of global LLGHG emissions under A1B are (IPCC WG III, 2000):

year:	1990	2020	2050	2100
fossil fuel CO_2, GtC/yr:	6.0	12.1	16.0	13.1
land use CO_2, GtC/yr:	1.1	0.5	0.4	0.4
SO_2, MtS/yr:	71	100	64	28
CH_4, MtCH_4/yr:	310	421	452	289

The IPCC attaches no likelihood or ethical value to its scenarios. They are conceived as options allowing forcing time-series of LLGHG to be estimated for input to models, whilst collectively taking account of known uncertainties about the future state of the world. No account was taken of 'unknown unknowns', including shocks such as might result from major wars, persistent economic depression, or widespread environmental collapse.

[5] From en.wikipedia.org/wiki/Special_Report_on_Emissions_Scenarios.

Under this A1B scenario, atmospheric CO_2 is predicted (as an average over many simulations) to reach 700ppm by 2100 (Meehl *et al.,* 2007). The most likely consequence is that mean planetary surface temperature in 2100 will be about 3.5°C warmer than that in 1900 or 2.5°C warmer than that in 2000.[6] However, as **Figure 4.11.** shows, there is some uncertainty in this prediction, the outcome of insufficient knowledge of some of the key processes and how to parameterise them in models.

Climate change impacts on ecosystem state and thus on services to human society. Given a schedule for warming, etc, coupled physical and ecological models might be used to estimate the impact on services, as discussed elsewhere in this chapter. However, each additional modelling step adds uncertainties, although what these are is much less well known than is the case for climate models. Finally, it can be envisaged, at least in principle, that a partial social system model could be used to propagate the ecological changes into economy and society (i.e. to effects on human welfare), including some direct feedbacks from the social system to the natural system as shown in **Figure 4.10.**

The socio-economic storylines developed for climate change studies have typically been constructed in relation to two main axes of societal variation. In the case of IPCC, one of these axes deals with psychosocial orientation to values relating to societal organisation. At one extreme lies individualism and a view that the market is the best way to allocate resources; at the other extreme is collectivism and environmentalism, with the recognition of both social and ecological interdependence. The second axis deals with the dominant scale of governance, from global to regional or local. Modelling techniques are beginning to be available that could allow natural resource management to be expressed as functions of these two state variables (see **Section 4.5.2.3**).

Thus, an algorithmic representation could in principle be made of a highly idealised social model, which could be forced by different socio-economic scenarios. In practice the complexity of the social system, and lack of knowledge about how to quantify key interactions, would likely make any predictions very uncertain indeed. Finally, adding to the difficulty of making an overall simulation model, are the feedback loops that exist in this SES. Three of them are shown explicitly, in (i) the response of climate to cumulative emissions of LLGHG, (ii) the response of society to changes in ecosystems providing services, and (iii) the response of global society to perceptions of climate change.

Given experience in developing Atmosphere-Ocean General Circulation Models (AOGCMs) and Earth System Models of Intermediate Complexity (EMICS) (see **Table 4.A3.**), it would seem that ensembles of social-ecological model able to make reliable simulations[7] on global and regional scales, would take decades of time and billions of dollars for development. The FP6 SPICOSA project found that models of intermediate complexity could be made relatively cheaply and quickly to help policy-makers in assessing solutions to specific environmental problems, such as eutrophication, or carrying capacity for shellfish (see **Box 4.3.**). Such task-based models, assembled from a toolbox of existing algorithms, could be used to apply the results of climate change simulations to particular ecosystem services in specified locations,

[6] Multimodel mean SAT warming (°C) for 2090-2099 compared with 1980-1999 are B1: +1.8 (1.1 - 2.9); B2: +2.4 (1.4 - 3.8); A1B: +2.8 (1.7 - 4.4); A1T: +2.4 (1.4 - 3.8); A2: +3.4 (2.0 - 5.4); A1F: +4.0 (2.4 - 6.4). The ranges in parenthesis are from -40% to +60% of the mean (Meehl *et al.*, 2007).

[7] The meaning of reliable' is discussed in **Section 4.5**. It is a particularly difficult concept when applied to simulations of scenarios, which may in practice never occur as postulated, and, furthermore, may be actively avoided when their consequences are appreciated. Roughly speaking, a reliable model is one that can accurately simulate historical change and that would prove to have simulated future change accurately, if the particular scenario were to take place exactly as described.

and could incorporate relevant specific aspects of socio-economic scenarios such as the schemes legally imposed for fisheries management. However, even such a relatively economical approach was beyond the budget and capability of the present study.

4.3.1.4 The Expert Workshop approach

Whereas the climate change simulations driven by the IPCC scenarios were achieved algorithmically, the scenarios were devised by a different method (IPCC WG III, 2000). The '*SRES writing team included more than 50 members from 18 countries who represent a broad range of scientific disciplines, regional backgrounds, and non-governmental organisations.'* Their core task was the *'formulation of four narrative scenario "storylines" to describe alternative futures'*; subsequently, the storylines and the consequent emissions schedules were subject to open and internal reviews. Our conscious human minds have very little algorithmic capacity; we make judgments in complex cases based on other ways of assessing evidence. This can be problematic in that it may lead to biased conclusions, and can fail completely when it comes to problems that have not previously been experienced, such as devising and applying socio-economic scenarios. However, structured methods have been devised to overcome biases and to aid thinking in new contexts. These include juries, councils of the wise, and the Delphi method. A key aspect of each of these is that individual validity claims[8] are tested by cross-examination, discussion or peer review.

As is widely admitted, future developments are not always correctly predicted by consensus of experts. For example, if Delphi panellists are misinformed, the use of Delphi may simply lend confidence to ignorance (Green *et al.*, 2007). Experts are necessarily strongly influenced by disciplinary paradigms which may turn out, as the world changes and knowledge evolves, to have been incorrect. Furthermore, in making holistic judgements about complex scenarios, experts move outside their specialised knowledge and may be biased by particular experiences or by a world-view created by media, education or belief-systems.

Nevertheless, models themselves depend on collective expertise and validation against past events to justify extrapolation to the future.[9] It is at least arguable that expert workshops, run according to Delphi principles, with opportunities to examine validity claims and re-assess initial assumptions, can provide a rough and ready estimate of future possibilities of equal reliability, but at much lower cost in cash and time, than may be obtained from complex social-ecological models. Of course, experts are not precluded from using model results, where available, as evidence.

4.3.2 Methods

On the basis of the arguments set out above, an expert workshop was convened in Edinburgh on April 18-19, 2013, to explore how marine ecosystem services might change between 2013 and 2060, given likely climate change and under five scenarios for socio-economic change. The scope of the exercise was defined as the UK's marine area but incorporating any necessary drivers beyond it, and a time horizon of

[8] Referring to 'validity claims' places the focus on statements and their authors, in contrast to reference to 'reliability' or 'accuracy' in models, which either assumes validation against observations, or the use in these models of well-tested hypotheses about processes in the natural or social worlds. Of course, the evidence advanced to justify a validity claim may include reference to observations or to well-tested theories.

[9] In **Section 5**, we discuss the virtues of mechanistic models - based on strongly tested theories - for extrapolation outside the domain of their validation, in preference to empirical (i.e. purely statistical) models.

2060. This is within the timeframe of the UK Office of Budget Responsibility's Fiscal Sustainability Report projections[10] for the next 50 years. A novel aspect of the workshop was to consider the effect of system shocks on the services.

4.3.2.1 The workshop

The workshop was attended by 26 persons with knowledge of marine ecosystems and their services, and with a willingness to engage in 'communicative action'[11] including the consideration and evaluation of validity claims. The participants included academics and stakeholders in environmental governmental organisations and NGOs; some also had expertise in facilitating workshops of this type. The working methods were those of 24-hour, 'mini-Delphi' process (Green *et al.*, 2007).[12] After lunch on the first day, participants were briefed on the workshop purpose and the scenarios, stimulated to discuss constraints on UK marine ecosystem services and trade-offs amongst these constraints, and then asked to complete forms assessing changes in services under a 'Baseline' scenario.

This was followed by discussions in two groups of the potential effect of environmental or socio-economic shocks on marine ecosystem services under this scenario. Participants then met for dinner and further informal discussion; the workshop reconvening the following morning to split into four groups, each charged with discussing and assessing service change under four variant scenarios. Further forms were completed relating to these scenarios. Finally, all participants were asked to re-assess service changes under the 'Baseline' scenario.

4.3.2.2 Geo-political regions

An earlier pilot study for the scenario exercise had shown that regional differences within Great Britain were thought to be significant. Thus, three geopolitical regions (**Figure 4.13.**) were identified on the basis of macro-economic drivers, underlying geology and coastal morphology, and marine ecohydrodynamics. These were used as the basis for assessment of service changes. [13] The regions differ in population density, wealth, type of coastal landscape, and in risk of sea-level rise (Lowe *et al.*, 2009), which is greatest in the south-east and least in the northwest. Participants were asked to make, where possible, different assessments for each region.

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[10] cdn.budgetresponsibility.independent.gov.uk/FSR2012WEB.pdf

[11] Communicative action, aimed at increasing mutual understanding of a topic, involves the making and hearing of 'discursively redeemable validity claims', and may be contrasted with strategic action aimed at achieving a successful outcome e.g. for the institution one represents (Habermas, 1984).

[12] In accordance with Delphi method practice, we do not name workshop participants. We are however grateful to them for their time and enthusiasm, and additionally to those who acted as rapporteurs.

[13] The basis of the distinctions in geology and geomorphology - well known to influence social history and geography - were first considered (in the present context) at a Valuing Nature Network workshop in Plymouth, 22-23 November 2012. Concerning the idea of ecohydrodynamics (a term first used in print by Jacques Nihoul of Liège University in 1986) see Tett *et al.* (2007). The argument is that light penetration and seasonal stratification regimes strongly influence the type of pelagic and benthic communities. The sea in the SE region in **Figure 4.3.3**. is typically well-mixed (as a result of tidal flows) and turbid. The sea in the other two regions tends to be seasonally stratified offshore, with a variety of regimes inshore. See Pingree & Griffiths (1978) for a map of mixed and stratified waters. There is a gradient of human influence (including nutrient loadings) from high in SE through W/NW to low Scotland, reflecting population density and agricultural intensity. As a specific example of this regionalisation, post-glacial fjords (locally, 'lochs' or 'firths') occur only in parts of the Scotland region.

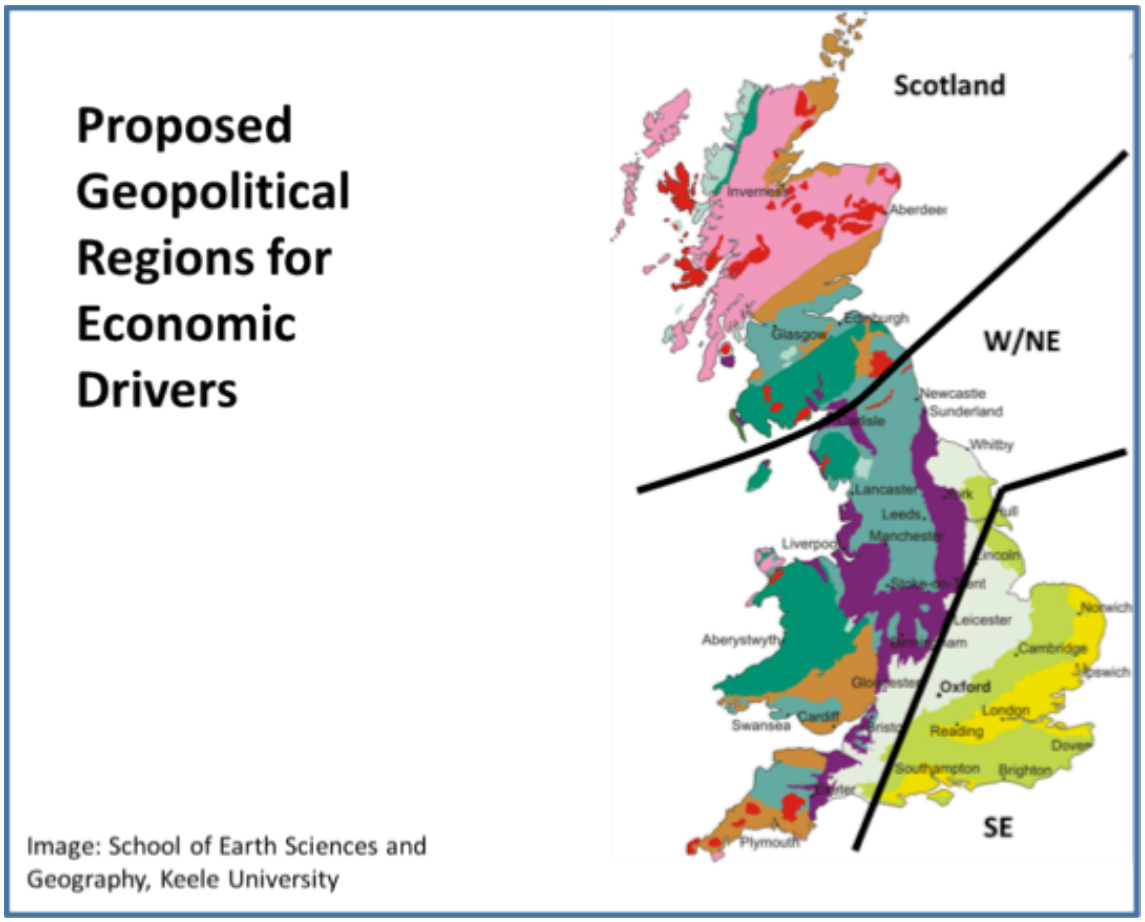


Figure 4.13. The geopolitical regions used for assessment. Source: School of Earth Sciences and Geography, Keele University.

4.3.2.3 The climate change scenario

For simplicity, single scenario for most aspects of climate change and ocean acidification was used, that predicted for typical emissions schedules by the models referenced in the UK Climate Projections for 2009.[14] UK coastal seas are expected to warm by 1-2°C by 2060, to become slightly fresher (due to increased rainfall and runoff), and to remain stratified for a few days longer in each year.[15]

[14] Acidification according to Turley *et al.* (2009). Jenkins *et al.* (2009) give a general briefing on projected UK climate change, and Lowe *et al.* (2009) specify marine and coastal details including sea-level rise and storm-surge risk. Lowe *et al.* mostly present results from a medium emissions scenario (the SRES A1B) except for sea-level rise (high emissions scenario, A1F). The UK projections take result from 'perturbed physics ensemble' (PPE - i.e. with parameter variation) runs of the HADCM3 AOGCM, which were used to force a higher resolution regional atmosphere climate model (HADRM3). Results were used to force models for waves, storm surges, and marine circulation.

[15] Sea-level rise projections were based on multiple model ensembles carried out for the IPCC 4th assessment. Changes from now until 2060 have been estimated from results reported in Lowe *et al.* (2009), chapter 6, for the period from 1961-1990 to 2070-2098. Stratification duration of course applies only to seasonally-stratifying waters.

Mean sea level rise was taken from a high emissions scenario as 3 mm/year (0.2m from 1990 until 2060) with 50% error bars.[16] Changes in land elevation increase the relative mean rise to 0.3m in the southeast and southwest and keep it at about 0.2m in the north and north-west. Although these are comparatively small increases, the main risk is from storm surges. These are harder to forecast, being comparatively rare events, but in the worst (simulated) case could combine with sea-level rise to add 1.5-2m to present-day astronomical high tide along much of the west coast of Britain, and in East Anglia and the Thames estuary.[17]

4.3.2.4 The socio-economic scenarios

The scenarios, devised originally for the ELME[18] project (Cooper *et al.,* 2008), are best thought of as regions within a socio-economic state space defined by axes for *governance* and *value-orientation*. The governance axis spans a range of societal organisation, from, at one end, strong interdependence on all levels, to, at the other end, a patchwork of locally autonomous communities. The value axis runs from strong consumerism, in which individual well-being needs are largely to be satisfied by through the impersonal use of money and markets, to strong communitarianism, in which well-being needs are mostly to be satisfied by interpersonal relationships and social provision. The five scenarios locate in different parts of this state space (**Figure 4.14.**), and the objective of the workshop is best seen as an attempt to assess marine environmental conditions in relation to each sector of this state space, rather than as attempting to predict futures as a function of socio-economic parameters. Four out of five of these scenarios correspond to those of Haines-Young *et al.* (2011) (**Table 4.1.**).The details of the scenarios that were provided to participants are given at the start of the relevant subsections in 'Results' (**Section 4.3.3**). They were largely taken from Cooper *et al.* (2008).

[16] Fig. 3.4 in Lowe *et al.* (2009).
[17] Interpolating between Fig. 4.8 and 4.9 in Lowe *et al.* (2009), for the upper bound of the PPE simulations, and for a projected once in 50 year surge.
[18] European Lifestyles and Marine Ecosystems (FP6 project, 2004-2007)

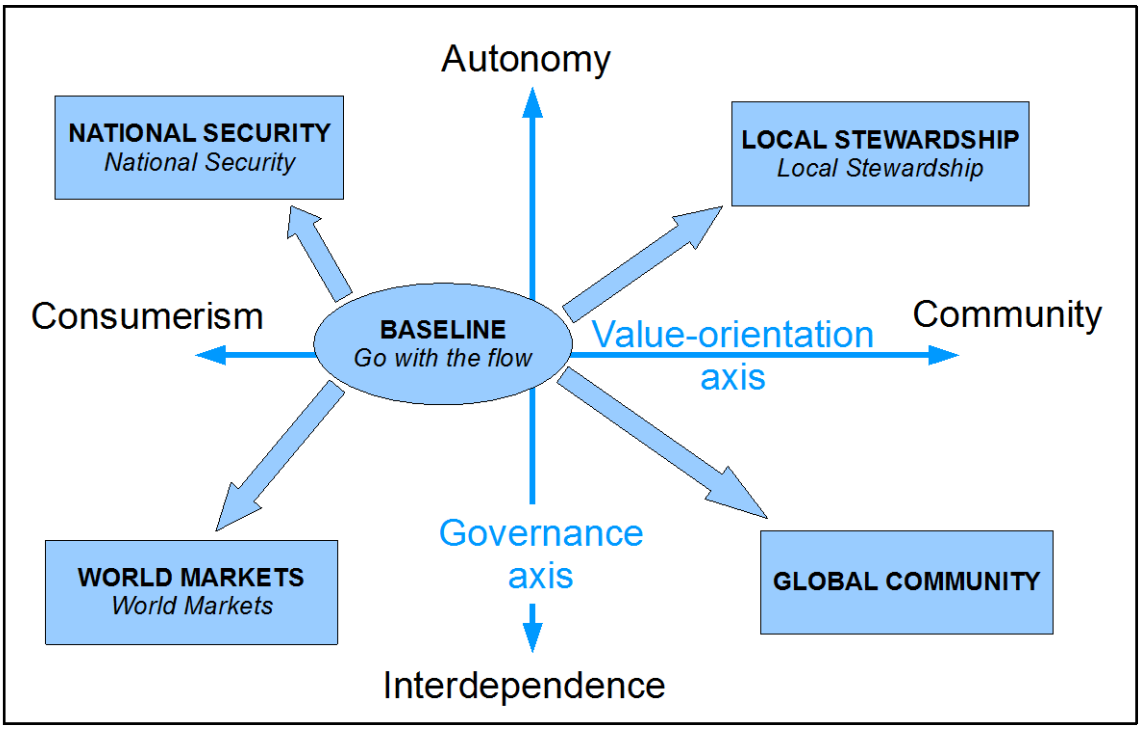


Figure 4.14. The scenarios plotted in a socio-economic state space. For example, 'Go with the Flow' refers to the corresponding scenarios of Haines-Young *et al.* (2011).

Table 4.1. Comparison of scenarios. Sources: ELME: Cooper *et al.* (2008); NEA: Haines-Young *et al.* (2011); MEA: Millennium Ecosystem Assessment (2005); WP4: scenarios used in this report.

	Scenario in outline	NEA 2011 name	WP 4 name	Other names
1	projection of present conditions and trends	Go with the Flow	**Baseline**	Business as Usual
2	national conservation funded from global markets	Green and Pleasant Land		
3	global free-market and environmental standards reconciled through valuing and nationally managing ecosystem services	Nature@Work		TechnoGarden (MEA)
4	strong subsidiarity, emphasis on environment and equity	Local Stewardship	**Local Stewardship**	Local Responsibility (ELME) Adapting Mosaic (MEA)
5	strong state and protection of national market economy	National Security	**National Security**	National Enterprise (ELME) Order from Strength (MEA)
6	global growth and free markets	World Markets	**World Markets**	
7	globalisation for equity and environment as well as market		**Global Community**	Global Orchestration (MEA)

4.3.2.5 Methods used to obtain and analyse data

Two sorts of data were obtained. Qualitative data took the form of narrative reports from subgroups, together with the comments recorded in the forms. The reports were used to prepare the descriptive accounts of the effects of scenarios, and the comments are summarised in the figures showing the results for each scenario. Inevitably, there was discussion concerning the desirability and feasibility of the world-views themselves, as well as their implications for ecosystem services, and this has been reflected in the narrative material in the 'Results' (see **Section 4.3.3**).

Quantitative data were obtained from the scores entered into the forms for each service: participants were asked to use a Likert-type 5-point scale (**Table 4.2.**) to assess the likely change in each service, in each geophysical region, under given socio-economic scenarios, assuming the pattern of climate change already described. Frequency distributions were derived from these scores, and are presented in **Figures** 4.**16-4.21**, scores such as 0/+ being allocated half to each category. These figures use the same layout, and list the same services, as the forms employed to collect data.

Table 4.2. Likert-type scale* used for assessment of changes in ecosystem services

score	interpretation in the workshop context	*value*
- -	strong opinion that all components will worsen	*-2*
-	expectation that at least some components will worsen	*-1*
0	no overall change expected	*0*
+	expectation that at least some components will improve	*+1*
+ +	strong opinion that all components will improve	*+2*

Note: *Likert (1932) proposed a 5-point scale for studying attitudes, with the subject being asked for agreement or disagreement with a statement.

An overall score was calculated for each service and region by (i) summing the product of number of scores in each category and the score-values given in the final column of **Table 4.2.**, (ii) dividing by the number of scores, and (iii) multiplying by 10 and rounding to give whole numbers between -20 (unanimous strong view that service will worse) and +20 (unanimous strong view that service will improve); the results for all scenarios are shown in **Figure 4.22.**, coded by colour from red (worsening) through yellow (no change) to green (improving).

4.3.3 Results

4.3.3.1 Baseline scenario

This is the result of projecting present trends from the present state of UK society and economy, albeit with the assumption of recovery from the post-2008 depression. Socio-economic changes (relative to present) were taken as those as shown in **Figure 4.15.** The following were also assumed:

- UK Seas will be spatially planned and that projected activities (e.g. areas licenced for renewables development, MPAs, decommissioning of North Sea oil, expansion of oil and gas extraction in deeper waters, some Carbon Capture Schemes) will continue; and

- existing policies, mostly resulting from EU drivers such as the WFD and the MSFD are fully implemented (UK Marine and Coastal Access Bill; Scottish Marine Act; multiple iterations of the EU CFP, increasing UK regionalisation).

DRIVER	BASELINE (Compared to present)		
	Scotland	W/NE	SE
DEMOGRAPHY			
Population size	+	+	++
Age distribution (proportion older)	+	+	+
Household numbers	+	+	++
ECONOMY			
Growth (in real GDP)	+	+	++
Structure (share of GVA) services/industry/agricult.	+/0/0	0/+/0	+/0/-
International trade in goods	+		
Debt/GDP ratio	+		
SECTORS			
Agricultural production	0	0	-
Energy production {consumption}	+/+	+/+	0/+
Wild fisheries production {capture effort}	0/-	0/0	0/-
Aquaculture production	+	0	0
Household consumption {Marine recreation}	0/+	0/+	+/+
Industry (includes service)	+	+	+
Tourism	++	+	++
Transport {terrestrial/maritime/air}	+/+/+		
DEVELOPMENT			
Coastal development	+	+	+
City development	+	+	++

Figure 4.15. Changes under the Baseline Scenario. W/NE: West/Northeast; SE: Southeast

As shown in **Figures 4.16.** and 4.**17.**, there was mild optimism about most services during the first round of scoring, which was tempered a little during the second round (as a result of the Delphi process). A key reason for this optimism was the view that national and regional environmental protection would become increasingly effective, supported by a public increasingly ready to accept proper costing of externalities. A regional trend is apparent, the result of lower population densities and greater recognition of the value of the environment (in itself and as a provider of services) in the north and west of Britain, in contrast to higher rates of population growth, urbanisation, and economic development in the south and east.

Final Ecosystem Service	SE England					NW/SW England + Wales					Scotland					Comments
	- -	-	0	+	+ +	- -	-	0	+	+ +	- -	-	0	+	+ +	
Wild fish/shellfish/seaweed	1	7	5½	6½			5½	8½	7			4	4½	11½	2	can MSY be acheived for all stocks as per CFP?; + because increase in less destructive types of fishing; compliance with Habitats directive should restrict fishing; change in species with warming; 0 because no capacity to increase
Cultured fish/shellfish/seaweed		1	6	13	1			3½	15½	1				9	11	constraint is investment for start-up/expansion; potential for offshore development in Scotland; increasing demand
Genetic resources		7	4	9			2½	6½	8	1		3½	6½	7	2	potential recognized but stocks decline; deep water species more important?; species loss because of temperature rise
Climate regulation		9	2	9	1		5	5	8	2		4	6½	7½	1	Tyndall centre expertise moved from SE to SW, influencing local regulation;
Natural hazard protection		7	3	5	6		1½	6½	10	1		1½	10½	7		- because of habitat decline; + because recognition of need for protection; isostatic rebound means less important in north?; global not national issue
Waste breakdown/ detoxification		5	5	9			1	9	7	1		1	9	8		worst in SE because of rate of population growth and urbanization; assume compliance with ND? assume that future waste is less toxic; assimilative capacity may fall with sea-level rise, especially given coastal squeeze
Meaningful places – Socially valued seascapes	1	4½	5	8½	2		3	4½	9½	5		4	3½	6½	6	preference for such seascapes, and visits to them, increase; increase with population growth, decrease with habitat decline; potential minor loss due to coastal development

General comments
if legislation successful most ecosystem services will increase

Initial scores (Thursday afternoon) for BASELINE scenario
median score

Figure 4.16. Changes under the Baseline Scenario. In this and similar diagrams, the numbers are of participants giving a particular score for the service and region. Where a borderline score (such as '0/+') was given, it has been counted half to each category. The median of the distribution is shown in grey. MSY: maximum sustainable yield; ND: Nitrates Directive.

Final Ecosystem Service	SE England					NW/SW England + Wales					Scotland					Comments
	- -	-	0	+	+ +	- -	-	0	+	+ +	- -	-	0	+	+ +	
Wild fish/shellfish/seaweed	1	10	8	2			7	5	9			5	3	9	4	acceptance of need to reduce fishing now for long-term gain; present policies inadequate; need better regulation, sustainable CFP and better evidence for management; overfishing by foreign (non UK or non-EU?) vessels; inshore fisheries easier to conserve
Cultured fish/shellfish/seaweed	1	2	11½	5½	1		2	8	9	2		2	1	7	11	SE England: small stock, some increase but lack of space; increases in production assume available assimilative/carrying capacity; need to distnguished seaweed cultivated for food or fuel; finfish farming constrained by feed supply and social acceptability; new species opportunities with higher temperatures
Genetic resources		6½	9½	5			6½	7½	7			5½	6½	8	1	0 & + result from improved protection: present policies will not slow loss of biodiversity; use of biodiversity likely to increase
Climate regulation		5½	8½	6	1		2½	11½	7			2½	8½	9	1	Small improvements in C sink management; C sink in newly created wetlands (defence against SLR); a global issue; -ve feedbacks likely as temperature increases
Natural hazard protection	1	6	4½	6½	3		3½	7	10½			1½	10½	9		Trend to value this over hard defences; need for more cost-effective (therefore, 'soft') defence against SLR; cease the coastal squeeze; SE will be better defended; local coastal partnerships in N and Scot.
Waste breakdown/ detoxification		4½	6	9½			1½	9	9½			1½	10	8½		Trend to value this over hard engineering; more waste as affluence increases. alternatively less discharge because of regulation; use wetlands created as C sinks and for defence; not a final ecosystem service; beneficial effects of WFD and RBMP
Meaningful places – Socially valued seascapes		5	7	7	1		1	8	9	2			5	12	3	increase in perceived value; increase in (scarcity) value; - in SE because economic and population drivers will dominate; meaningful landscapes may become less accessible
Recreation			2	1	1			3	1			1		2	1	

Final scores (Friday morning) for BASELINE scenario
median score

General comments

Culture shift in SE more difficult to acheive than in other regions; Scotland has more potential for improvement due to low population densities and stronger tradition of coastal/marine management

Figure 4.17. Baseline scenario, final scoring. RBMP: River Basin Management Plans.

4.3.3.2 Shocks to the baseline scenario

Physical and ecological shocks to UK marine ecosystems

Several sorts of physical and ecological shocks were considered[19]: a storm surge sufficient to overtop the Thames barrier; a 6 months period of reduced light and sea-surface heating resulting from a volcanic eruption on Iceland; blooms of an invasive species comparable to *Mnemiopsis* in the Black Sea; an extreme summer resulting in sub-thermocline de-oxygenation over large areas of coastal sea. In the group's view, most marine and coastal ecosystems remain sufficiently resilient to recover from such shocks, or in ecological terms, *pulse* disturbances, within a few years.[20] This resilience arises partly from the biological community and partly from the open and well-flushed nature of the seas around the UK. It is possible that such a shock might cause an ecosystem to shift from one regime to another, but the existing ecological evidence suggests that this is unlikely, because it is sustained, chronic, or in ecological terms *press*, disturbances that are more likely to have such consequences. Certain sorts of shock, such as coastal flooding, might have long-term consequences for the integrity of coastal freshwater wetlands and the services they provide. Other shocks might impact directly on certain services, for example on aquaculture, but their long-term impact would depend on their effect on the socio-economic rather than the ecological system. Finally, such shocks might have ecologically beneficial effects if they changed human perceptions of the environment and thus drivers of change. For example it might be decided to accept flooded areas as part of managed realignment of the coast, so diminishing the 'coastal squeeze' which greatly weakens the ability of littoral and supra-littoral communities to move and adapt to sea level change.

Political and Economic shocks

A complete breakup of the EU was considered unlikely; more realistic possibilities included failure of some EU member states with greater centralisation. The break-up of the UK was another possible shock. In either case it was thought that there would be minimal long-term disturbance of ecosystem services from those expected under the Baseline scenario.

The economic shock considered was that of a recession more severe than that experienced by the UK since 2008, perhaps accompanied by significant financial collapse of the state, and lasting for a substantial part of the period until 2060. The likelihood would be that an impoverished government

[19] The shocks that we examined were intended to be plausible if, hopefully, unlikely. The storm surge case corresponds to the upper end of the sea-level and surge increase range supposed in the 'extreme' H++ scenario of Lowe *et al.* (2009), combined with a lag in upgrading Thames estuary defenses. The main widespread effects of the 2010 eruption by Eyjafjalla-jökull were those on air transport, but the 1883 eruption of Laki over six months was followed by a deterioration in the climate of Europe, with cooler and wetter summers, and it has been suggested that the resulting crop failures led to the French Revolution of 1789-90 (Bressan, 2013). Clearly, such shocks can significantly perturb socio-political systems and, as exemplified by the levee breach in New Orleans during hurricane Katrina in 2005 (Vigdor, 2008), cause loss of life, great disruption of society, and massive loss of property. However, workshop participant were not asked to consider the socio-economic impacts, but the effects on marine ecosystem services, for example by the pollutants and debris washed down from flooded industrial sites around the Thames estuary.

[20] The distinction between *pulse* and *press* perturbations was introduced by Bender *et al.* (1984): a '*pulse* perturbation is a relatively instantaneous alteration of species numbers, after which the system is studied as it "relaxes" back to its previous equilibrium state. A *press* perturbation is a sustained alteration of species densities (often a complete elimination of particular species): it is maintained until the unperturbed species reach a new equilibrium.'

could not afford to enforce statutory protections of the marine environment, and thus that there would be increasing press disturbances of marine ecosystems through over-exploitation of services. The economic shock might however lead to a significant change in society, perhaps to one of the two 'green' scenarios and a stable zero growth economy.

4.3.3.3 National Security scenario

Description
Values & Policy: Individualistic, highly personal consumption, low taxes, market-based, but strong commitment to national culture and interests. Little concern for social equity or environmental protection. Sovereignty retained or taken back to national level. Erosion of EU, and protectionist measures weaken WTO.

Demography: Population affected by little inward migration and relatively low birth rates, although age distribution balanced to some degree by diminished longevity. Migration to internal growth 'hot spots' and average household size stable, but with household numbers increasing more slowly than under Baseline.

Economy: Priority of growth undermined by protectionist policies. Focus on meeting internal demand and security of supply. Trade diminished within EU but not as much as extra-EU. Considerable variation in regional development.

Assessment
Under the National Security scenario, the UK has taken a protectionist stance and has withdrawn from groups that are perceived to undermine its sovereignty. Thus it has left the EU and revoked national transposition of the CFP, the MSFD, Birds and Habitats Directive, etc. It continues to be a member of OSPAR, ICES and the International Maritime Organisation and has negotiated a complex series of bilateral agreements with its neighbours. The country has to pay huge attention to self-reliance for energy supply (nuclear, coal and deeper sea and Falkland oil) and spend increased amounts of money on protecting its borders and trade (it has very strong limits on immigration and the trade barriers have increased pressure from smuggling). The welfare state and environment have received much less state support and it is difficult to finance innovation. The marine biotech industry has stagnated or joined the ‘brain drain’. There are strong regulations to protect property rights and this has extended to marine property where the Crown Estate has become the de-facto regulator. With increased domestic tourism, landscape values are paramount (thought there is tension with weakened regulations on pollution control). The renewables industry has virtually disappeared. The National Trusts, English Heritage, Historic Scotland, etc. are more important than Natural England, EA, SEPA and other environmental bodies (mostly amalgamated). Environmental protection and planning is reactive rather than proactive.

Under this scenario, fisheries management went through cycles of boom and bust as bilateral agreements with neighbours were ineffective and effort controls crumbled. The difficult financial situation however, eventually led to the removal of all subsidies and this, combined with fuel price hikes, led to major bankruptcies and reduced effort. Franchising of rights to fishing companies led to improved stock management and the franchises agreed voluntary agreements with neighbours, though effort was generally beyond maximum sustainable yield. Aquaculture only developed in the context of the ‘luxury goods and exports’ market (mainly salmon) but warmer temperatures caused the spread of Pacific oysters which became popular for local prospectors.

Sea defences became increasingly expensive due to sea level rise. Only valuable assets (such as London's commercial district) could be properly protected and other coastal areas were lost during locally catastrophic 'un-managed' realignments'. Pollution control laws remained at about the same level as 2013; no significant new measures were developed; compliance declined; and there were increasing problems with cumulative impacts. Feedback from recreational users through strong local councils and landowner associations maintained some protection for beaches and bathing waters.

Group participants were generally pessimistic in their scoring of ecosystem services, expecting most to decline (**Figure 4.18.**) the exceptions being fisheries (as discussed above) and socially valued landscapes. The latter increase reflects a greater pride the national countryside and the increase in domestic tourism.

Final Ecosystem Service	SE England					NW/SW England + Wales					Scotland					Comments
	- -	-	0	+	+ +	- -	-	0	+	+ +	- -	-	0	+	+ +	
Wild fish/shellfish/seaweed				6					6					6		better fisheries management
Cultured fish/shellfish/seaweed		1	5					5	1			1	1	4		luxury goods for export (Scotland)
Genetic resources		3½	2½			1	2½	2½			1	2½	2½			no MPA
Climate regulation		4	1	1			4½	½	1			4½	1½			less marine protection
Natural hazard protection	2	2½	½	1			4½	½	1			2½	3½			unmanaged re-alignment
Waste breakdown/ detoxification	1	3½	1½			1	2	3				3	3			- due to high population increase in SE; greater demands, reduced assimilative capacity; lowered compliance; 0 due to greater pressure to preserve environment for tourism
Meaningful places – Socially valued seascapes		1		4	1		1	1	4			1		5		+ due to more domestic tourism; - due to unregulated development

scores (Friday morning) for NATIONAL SECURITY scenario

median score

Figure 4.18. Scoring for the National Security scenario.

4.3.3.4 World Markets scenario

Description
Values & Policy: Libertarian, techno-centric materialist consumerism. Presumption in favour of market provision. Growth more important than social equity, with environmental policy limited to correction and support of the market. Increased global interdependence and governance, through WTO and multinational corporations. Corporate governance starts to displace government. Policy determined at regional trading bloc and international level. Rapid enlargement of EU.
Demography: Population growth slows overall but migration increases to meet demand for labour and reduces proportion of older people. Growth uneven across regions. Smaller and more numerous households.
Economy: Rapid growth, with dismantling of trade barriers increasing intra- and extra- EU trade. Service sector dominates others, with decline of agriculture and manufacturing. Benefits of growth spread to some extent through 'spill over' effects.

Assessment
It was concluded that outcomes depended crucially upon the ability of governing bodies to correct for externalities. It might be that an international body would have sufficient authority to impose strong environmental regulation/certification, based upon the collective understanding that continued growth is dependent upon functioning ecosystems. It is not inconceivable that there would be a rocky transition path to this eventuality, with significant environmental degradation in the medium-term before the wider community – including financiers and investors – realised that this degradation was increasingly having a negative impact upon profit potential, and consequently ensured the business world backed greater regulation. A fundamental element of such regulation would be a fully-operational market for carbon. In this scenario, all natural assets would be privatised and would be managed on the basis of property rights. Fish stocks, for example, would be managed by a global system of tradable quotas, very likely leading to greater consolidation of fleets and enhancement of profitability. The owners of these (now private) assets would have a direct incentive to manage them sustainably, including their supporting ecosystems.

On the other hand, it could be argued that what has just been described is closer to the Global Community scenario, and that, in the World Market scenario, society is too myopic to take any such long-term action. In this case, a failure at the international level to manage externalities could lead to ‘mega-death’. The key driver of this would likely be climate change, with temperature rises of up to 6°C. Should this happen, large global shifts in population would take place as increasing swathes of land (Australia, Bangladesh, much of Africa) become uninhabitable due to either flooding or drought. The resulting pressure that this would place upon remaining natural resources would lead to an increasing downward spiral in the most impacted countries and potential international conflicts, particularly over scarce resources such as oil. The only identifiable brake on such a course of events would be the insurance market via increasingly expensive insurance premiums as risk increased. Within the UK, the south-east is likely to be most detrimentally impacted. **Figure 4.19.** reflects the scoring of the World Markets scenario.

Final Ecosystem Service	SE England					NW/SW England + Wales					Scotland					Comments
	- -	-	0	+	+ +	- -	-	0	+	+ +	- -	-	0	+	+ +	
Wild fish/shellfish/seaweed	4	1				4	1				2	2½	½			overfishing; climate change effects
Cultured fish/shellfish/seaweed		1	2½	1½					2	3				1	4	acidification effects shellfish; need to import fish feed
Genetic resources	2	3				2	3				2	3				
Climate regulation	3	2				4	1				4	1				no global management of climate change
Natural hazard protection		5					1	4				1	4			SE England has increase in man-made protection
Waste breakdown/ detoxification	2	2	1				2½	2	½			2½	2	½		
Meaningful places – Socially valued seascapes	4	1				1	1½	2½			1	2½	1½			more people and agriculture

scores (Friday morning) for WORLD MARKETS scenario
median score

Figure 4.19. Scoring for the World Markets scenario.

4.3.3.5 Global Community scenario

Description
Values & Policy: Communitarian, with internationalist values and increasing globalisation of governance systems to deal with global, interconnected, problems. Balancing of economic, social and environmental welfare, with preference for latter and willingness to accept high tax levels. Policy co-ordinated at EU and international level, but implemented at local level. EU more centralised, with less regional autonomy, and slower expansion. Environmental policy expands across policy sectors and is prioritised. Powerful, green, WTO favours environmental protection in trade disputes.

Demography: Low birth rates offset by migration increases to meet demand for labour, with some increase in average age but relatively static distribution. Household size declines slowly, and numbers grow at historic rates.

Economy: Growth constrained by tax levels and social and environmental objectives. Shift to services is slower than in Baseline. Growth in intra- and extra-EU trade, but with some inhibition through 'footprint' concerns. Development evenly distributed, though with some transitional variations.

Assessment
In this world the high level policy goal is ‘strong’ sustainability, underpinned by a macroeconomic strategy based on a ‘slow’ growth philosophy and practice. There is emphasis on maintaining and/or improving overall wellbeing and the stock of wealth (i.e. discounted present value of a future consumption stream anchored to all four forms of capital – physical, human, natural and social). Population growth is being stabilised. The global economic system and network of interdependencies is being radically reformed. Remits of some international institutions are being re-orientated in order to better enable the ‘slow’ growth strategy. For example, the WTO has had its 'fair trade' brief expanded to include environmental sustainability concerns. Banks have had their retail and investment activities completely separated. A ‘Tobin’ tax is in force internationally which is constraining international speculation and its destabilisation (‘bubble effects’) of financial, energy, property and commodity markets. The World Bank and IMF have been assigned a stronger regulatory role covering both financial and environmental management. Natural capital and its contribution to ‘wealth’ is now formally part of the national/international income/wealth accounting practice.

Overall, a much more extensive and interventionist regulatory regime is in place; a stricter and ‘smarter’ set of policy measures are operating at the international and national scales. International environmental agreements have been negotiated, agreed and are being rigorously enforced; a climate agreement is in place and is on course to meet a 2-3° warming target; and the law of the Sea Convention has been given strong legal ‘teeth’, alongside integrated coastal management and other marine related governance.

The UK is following a ‘green’ growth strategy with an emphasis on innovation and investment in resource saving and recycling technologies, covering, energy, water, waste and other raw materials. Public transport infrastructure investment is favoured over private transport. Supply chains are being reorganised to make them as short as is feasible. Product differentiation and persuasive advertising are discouraged. Resource exploitation is constrained by the precautionary and 'polluter pays' principles, and risk minimisation rules have precluded exploitation of ‘fragile’ areas such as the Arctic. Such areas are zoned and kept clear of all activities except scientific research.

The state is intervening to try to redistribute income and wealth more equally i.e. actively seeking to reduce the gap between the top and the bottom of the income distribution, through progressive taxation and other fiscal means. Attitudinal and behavioural change is evident across both civil society and the business communities. Social networks are encouraging new social norms focused on reflexive citizenship and corporate responsibility and ethics. A culture involving the maximisation of short term desires and profits is being replaced by a culture favouring longer term needs and 'average' rates of return. Fair compensation and equity have been adopted as principles to be applied in any significant resource conflict/trade–off contexts. This scenario, it was concluded, was likely to led to increases in the sustainability of most ecosystem services, as shown in **Figure** 4.**20.**

Final Ecosystem Service	SE England					NW/SW England + Wales					Scotland					Comments
	--	-	0	+	++	--	-	0	+	++	--	-	0	+	++	
Wild fish/shellfish/seaweed					4					4					4	stocks exploited at MSY, low effort, small economic units; global perspective on stock constraints; equitability amongst sectors; fish preferred to meat as protein source
Cultured fish/shellfish/seaweed			3	1					4					4		requires development of sustainable farming: multitrophic, regulation of antibiotic use and feed sourcing; expansion limited by conservation zoning and concerns about wild population genetics
Genetic resources			4					4					4			interpreted as ecosystems, habitats and species diversity
Climate regulation				1	3				4					4		more 'soft' engineering of coastlines (more saltmarshes, mudflats) seagrass beds recovering with better water quality; increased carbon storage (including in wetlands), microclimate regulation;
Natural hazard protection				1	3				4					4		ICM including above, with managed realignment, beach and dune augmentation; Reduced coastal squeeze and lower maintenance costs for 'hard' defences
Waste breakdown/ detoxification				4					4					4		improvements leading to better water quality, helped by overall reduction in waste generation in catchments, on coasts and at sea
Meaningful places – Socially valued seascapes				4					4					4		greater appreciation of these cultural asset values; switch from international to domestic tourism, with restrictions on some damaging coastal recreation

(largely consensual) scores (Friday morning) for GLOBAL COMMUNITY scenario

median score

Figure 4.20. Scoring for the Global Community scenario. MSY: maximum sustainable yield; ICM: Integrated Coastal Management.

4.3.3.6 Local Stewardship scenario

Description

Values & Policy: Communitarian, co-operative self-reliance. High levels of public services funded by high local taxation. Strong emphasis on social equity and environmental protection at the local level. Local government replaces national and supra-national governance. EU becomes more diverse with regional autonomy and fragmented policy.

Demography: Population size stable, but relatively low birth rates and increased public health provision increases average age. General migration away from cities, with household size increases and household number reductions.

Economy: Slow growth, exacerbated by tax levels, with increases in smaller scale production. Trade greatly diminished, but with some preference for intra-EU over external trade. Growth more even across communities.

Assessment

Under this scenario, local stewardship has proven to be effective in promoting improved conservation of coastal and near-shore marine ecosystems and sustainable use of the resources they generate. However, local stewardship initiatives work best where there is an enabling environment where national policies, technical support, and willingness of national sectoral agencies to respect and work in concert with local initiatives. This has to some extent been provided by UK federal and EU legislation, along with increased use of the principal of subsidiarity. Decoupling of terrestrial, coastal and marine systems management has been partly overcome by the application of integrated EU Directives, such as the River Basin Management approach of the WFD.

Local Stewardship approaches have also proven vulnerable to strong external forces beyond their control. For example, local community management of fisheries has encountered difficulties offshore, where communities do not have the resources to implement fisheries management measures and impose them on out-of-area exploiters. The local effects of globally generated climate change and sea-level rise are another example.

There are regional differences in the capacity and effectiveness of local stewardship approaches in helping to solve regional and national ecosystem management issues. Regions such as Scotland may have increased capacity to expand coastal and nearshore production of marine based protein to help feed the more densely populated areas of England. Likewise, parts of England have the climate and soils that can produce enhanced yields of carbohydrates to help meet the needs of people in Scotland. However, given the differences in population pressures and differing economic foci of the human resources between regions, there will be differing interests in and ability to foster local stewardship. For example, Financial Services in London and the southeast of England currently dominate the UK economy. The Global Markets outlooks involved in these activities may counteract the effectiveness of local stewardship in improving the management of ecosystems and maintaining the quality and quantity of renewable resource flows.

The effect of these reservations (about the tension between local stewardship and the need for national, continental or global scale regulation) is reflected in the range of scoring in **Figure 4.21.**, even if the majority of participants were optimistic about outcomes.

Final Ecosystem Service	SE England					NW/SW England + Wales					Scotland					Comments
	- -	-	0	+	+ +	- -	-	0	+	+ +	- -	-	0	+	+ +	
Wild fish/shellfish/seaweed			1	3	2				4	2				4	2	better local management and self-policing; co-operation amongst coastal communities, backed up by national and international action
Cultured fish/shellfish/seaweed				6			½	½	4	1		½	½	1	4	SE England increase for local use; seaweed used for food and fuel
Genetic resources			2½	3½				1½	4½				1½	4½		biodiversity prospecting, valuation and conservation
Climate regulation		3	2	1			3	2	1			2½	2	1½		global pressure on a localized system; vulnerable to external forces; low local influence but greater responsibility
Natural hazard protection		1½		4½			½	1	4½			½	1	4½		local management to fit local needs (better understood locally); vulnerable to external forces
Waste breakdown/ detoxification		1	2	2	1		1	2	2	2		1	2	2	1	local management to fit local needs; vulnerable to external forces (which will need higher-level regulation) - desirability of RBM
Meaningful places – Socially valued seascapes			1	5				1	5				1	5		generally more highly valued; strong emphasis on protection

scores (Friday morning) for LOCAL STEWARDSHIP scenario

median score

Figure 4.21. Scoring for Local Stewardship scenario. RBM: River Basin Management.

4.3.4 Discussion

4.3.4.1 The scenarios

As considered in the Introduction, the use of scenarios in strategic planning is well-established in the military, commerce and environmental impact assessment. The IPCC has presented scenarios both for the generation of GHGs (and thus climate change) and for determining the impact of climate change on society. UKCIP (2000) applied the latter to the UK. The MEA (2003; 2005) used scenarios to explore a range of interactions between humans and ecosystem sustainability, and the UK NEA (Haines-Young *et al.*, 2011) applied this to the UK. Broadly speaking, it is possible to distinguish two main uses of scenarios: (i) to identify possible futures, elucidate preferences, and decide on actions to make some futures more likely than others; (ii) to explore possible environmental states and the consequences of these for the target system, so that, the former being taken as given, the latter can be exploited or ameliorated. Haines-Young *et al.* (2011) point out that although scenarios (and their consequences) are often seen as *products* of a formulation exercise, it may be more productive to understand them in terms of *process:* which is to say that it is the discussion amongst stakeholders that is the most valuable result.

Our workshop is thus best seen as an exploratory exercise in which *process* (discussion of options) was as important as *product* (prediction of future state). We asked participants to enter imaginatively into the world-view of a given scenario in order to deduce its impacts on services. Ideally, this required participants to take on the attitudes to environment that were part of the scenario and to seek methods for sustaining services in that world, even if the steering institutions in the scenario gave lower priority to such sustainability. It might be said that we were asking participants, even when they took a bleak view of the scenario, to think of ways to get the best outcome in this undesirable world.

As Charles Dickens implied, when he wrote that 'it was the best of times; it was the worst of times',[21] there is no state of society that is not a mixture of good and bad. Similarly, so long as humans remain numerous in the real world, there are no foreseeable environmental utopias, but we should expect no complete dystopias either. Eleanor Ostrom and colleagues have argued that there are no panaceas for environmental problems, no single recipes for ways in which society should be organised so as to move towards sustainability (Ostrom, 2005; 2007; 2009). Discussions of scenarios can also be seen as creative ways to identify particular solutions to environmental challenges, and some of these solutions might emerge in responses to scenarios that initially seem to be most unpromising for environmental sustainability.

4.3.4.2 Opinions about change under each scenario

Despite considerable differences of individual opinion concerning the scores for change in particular ecosystem services in a given scenario (**Figures 4.16.-4.21.**), there was a clear outcome from the workshop, shown in **Figure 4.22.**

[21] The opening words of 'A Tale of Two Cities'

Scenario	Baseline (1)			Baseline (2)			National Security			World Markets			Global Community			Local Stewardship		
Final Ecosystem Service / Region	SEE	N&W &W	SC	SEE	N&W &W	SC	SEE	N&W &W	SC	SEE	N&W &W	SC	SEE	N&W &W	SC	SEE	N&W &W	SC
Wild fish/ shelfish/seaweed	-1	1	5	-5	1	6	1	1	1	-18	-18	-13	20	20	20	12	13	13
Cultured fish/ shelfish/seaweed	7	9	16	2	5	13	-2	2	5	1	16	18	3	10	10	10	9	14
Genetic resources	1	4	4	-1	0	2	-6	10	-8	-14	-14	-14	0	0	0	6	8	8
Climate regulation	1	4	3	1	2	4	-5	2	-8	-16	-18	-18	18	10	10	-3	-3	-2
Natural hazard protection	5	6	3	2	3	4	-9	-6	-4	-10	2	-2	18	10	10	5	7	7
Waste breakdown/ detoxification	2	4	4	3	4	4	-9	-7	-5	-12	-4	-4	1	10	10	5	8	5
Meaningful places - Socially valued seascapes	3	8	7	2	6	9	8	5	7	-18	-5	-9	1	10	10	8	8	8

Figure 4.22. Expert opinions about the relative change in delivery of marine ecosystem services under five different scenarios in three British sub-regions: South East England (SEE); North and West England and Wales (NWW); and Scotland (S). Green is positive (good, max. score +20); white is relatively little change; and orange is negative (bad, max score -20). The Go with the Flow scenario was assessed by the group before (Baseline 1) and after (Baseline 2) the deliberations. Since marine systems are open and heavily influenced by global and regional policies, a Global Community scenario was devised and tested (emphasising wider international factors and increased globalisation of governance in the maritime environment), rather than the Nature@Work and Green and Pleasant Land scenarios designed for terrestrial environments. All other scenarios are those from the UK NEA.

Opinion was that the World Markets and, to a lesser extent, the National Security scenarios would likely lead to strong impairments in most marine and coastal ecosystem service, whereas the Global Community and Local Stewardship scenarios would lead to improvements. Explanations involved the priority given to environmental sustainability in the last two scenarios, the primacy of the market in World Markets, and the reactive and partial nature of governance in National Security. There was fair consistency amongst the two scorings of the Baseline scenario, and the median opinion was slightly optimistic for most services. The key explanatory factor in this case was the view that current environmental legislation, mostly transpositions of EU directives, would be fully implemented and enforced. A minority opinion expected economic drivers to prove stronger than the will to protect the environment. Regional differences are expected under all scenarios, typically the result of a gradient from the southeast of England (where population and consequent pressures are highest) to Scotland (with mostly lower pressures and an environment suitable for aquaculture).

The comments provided by workshop participants suggest ways in which present and near-future management practices could be modified to improve sustainability of ecosystem services without requiring substantial changes in societal organisation. Drawing on the highest scoring scenario, that for 'Global Community' (**Figure 4.20.**) suggests, for example, the benefits of 'soft engineering' of coastlines and of multitrophic aquaculture. Such technologies might be of immediate value as well as providing resilience against climate change. As discussed in **Section 4.5**, purpose-specific models could be used to explore their costs and benefits.

Both discussions concerning shocks to the Baseline scenario led to the conclusion that 'what does not destroy us, makes us strong'. [22] Marine ecosystems were seen as resilient against pulsed physical or ecological disturbance, and, the UK socio-economic system was seen as similarly resilient against foreseeable political or economic shocks.[23] In the medium term, such shocks may lead to the development of greater resilience, exemplified by greater use of managed coastal re-alignment after a Thames Barrier overtopping. In ecosystem theory (Bender *et al.,* 1984) there is a distinction between short-term 'pulse' disturbances and long-term or chronic 'press disturbances', the latter capable of eroding system resilience until (with or without a trigger) the system collapses or switches to a new configuration. Such distinction might also apply to socio-economic systems, the 'press disturbances' being those that shifted them to a different location in the state space of **Figure 4.14.**, i.e. into a different scenario.

[22] Commonly ascribed to Nietzsche in 'Götzen-Dämmerung, oder, Wie man mit dem Hammer philosophiert. C. G. Naumann: Liepzig, 1889. However, Nietzche wrote in the singular: 'Was mich nicht umbringt, mach mich stärker'. According to http://en.wikipedia.org/wiki/Twilight_of_the_Idols, G. Gordon Liddy, former assistant to US President Nixon, paraphrased it as 'that which does not kill us makes us stronger'. The relevant point here is that crises in complex systems, such as minds, societies or ecosystems, can be disastrous, or they can lead to system reconfiguration and recovery in a more resilient condition. See for example Gunderson & Holling (2002)

[23] As already footnoted, our focus was mainly on the response of marine ecosystems to these shocks. An event such as Thames Barrier overtopping would be catastrophic for many citizens, and although the socio-economic system would recover, the costs might fall unequally across social groups, as occurred when New Orleans was flooded by hurricane Katrina in 2005. Our optimism about the resilience of the UK socio-economic system is based on the view of effective multi-tier governance (at local, regional and national levels). The market economy might be less resilient, due to 'just-in-time' supply chains and the possibility of bank collapse.

4.3.4.3 A Delphi process?

The results in **Figure 4.22.** derive from the opinions of a particular group of workshop participants who were able to bring to bear their expert knowledge of marine ecosystems and their interactions with human society. How far can the results be considered reliable either as predictors of future conditions ('what will happen') or as explorations of social-ecological state space ('what would or could happen if ...')? Three sorts of difficulty might lead to erroneous conclusions.

1. Strategic rather than communicative action. Participants might come to the workshop with an intention of influencing its conclusions so as to favour a particular sector in society. This is 'strategic action' in Habermas' (1984) typology of interactions. However, the workshop was organised so as to favour 'communicative action', in which participants discursively challenge each other's' 'validity claims', so arriving at greater understanding of, and respect for, each other's' positions and views, with an outcome that accurately reflected the sum of participants' knowledge of the physical and socio-economic worlds.
2. Different understandings: for example, of the distinction between an ecosystem's capacity to supply a service versus the extent to which the service is used;[24] or of the understanding of 'meaningful place': although some coastal landscape might be lost to development, some of the remainder might become more highly valued either because of scarcity or change in appreciation-values. Understanding of differing interpretations increased as discussion continued, and this helps to explain some of the range of scores shown for particular services in **Figures 4.16-4.21.**
3. Biased worldview. It is likely that participants had preferences for certain of the socio-economic scenarios and that, given the nature of the participants as a group, that there was an overall bias towards or against identifying with any given scenario. But how far might that be expected to prejudice assessment of the effects of a particular scenario on ecosystem services?

All persons are in immediate contact with a social 'lifeworld' (Habermas, 1984) and a directly experienced physical world,[25] and ideas or theories about these worlds are open to straight-forward test and revision. This is not the case for ideas about world-views that are largely formed, in the modern world, by institutions including education and the media. Such views do not guarantee an accurate conceptual model of the real social and physical world. Assessing the likely impact of complicated social changes on complex natural systems, as we required of workshop participants, is likely to be influenced by their world-views as much as by the specialised knowledge that experts have of the real world within their domains of expertise.[26] In principle, assembling a sufficiently wide range of expertise and experience, including some contrary points of view, and allowing sufficient time for 'validity claims' to be understood, examined, and tested discursively, should render the constructed world transparent and allow expert judgment to assess real-world ecosystem changes with accuracy. Because our discussions were time-limited, it was not possible fully to explore and test each participant's understanding of the causal chain from socio-economic Drivers through Pressures to changes in ecosystem State and Services.

[24] An example is the expected increase in fin-fish aquaculture in Scottish waters as a result of Scottish government policy; it is considered that there is a reserve of untapped environmental capacity to assimilate farm wastes.

[25] 'Physical' is used here as shorthand for the material world, 'world 1' of Popper (1978), and refers to chemical and biological as well as mechanical processes and entities.

[26] This is to suggest that experts can 'see through' the media-constructed world to the real physical or social world, and test and revise their understandings by direct (even if instrument-mediated) observation of a limited part of the real world in the same way as they directly experience their immediate environment. Or, at least, they read peer-reviewed papers about other experts' direct observations and experiments. Even experts, however, are not immune to the world-view created by their current disciplinary paradigms.

Furthermore, it is unclear how far participants were able to enter imaginatively into the world-views implied by scenarios such as Global Markets or National Security, which were distinctly less attractive to most of us.

These three points relate to the reliability of a Delphi process, a topic that has been discussed at length in the literature from both practical (e.g. Richey *et al.*, 1985; Green *et al.*, 2007) and epistemological standpoints (e.g. Mitroff & Turoff, 2002). Insofar as such a process is deemed to be reliable if it moves towards an expert consensus, the results (**Figures 4.16. and 4.17.**) of repeated soliciting of views on the Baseline scenario suggest that views were fairly robust against both discussion and overnight reflection and thus that the conclusions reached about this scenario (at least) reliably reflect challenged opinion.

4.3.4.4 Scenarios

The scenarios of Haines-Young *et al.* (2011) were devised to help in answering questions raised by stakeholders about the ways in which the effects of climate change on ecosystem services might be influenced by imaginable differences in society and economy, and they were presented in terms of five dimensions involving society, economy and the physical world. These dimensions can be seen as variables that define the state of a social-ecological system, and are so presented in **Table 4.3.** In contrast, the ELME-derived scenarios used in the present exercise related to a social system defined by only two state variables. Nevertheless, most of the resulting scenarios are similar between the sets. The main divergence between the two sets (**Figure 4.14)** is in the 'Global Community' scenario of the present exercise, which does not correspond to either of the 'Green & Pleasant Land' or 'Nature@Work' scenarios of Haines-Young *et al.* **Figure 4.23.** builds on **Figure 4.14.**, and attempts to locate all the UK NEA 2011 scenarios, as well as ours, in a two-dimensional state space.

Table 4.3. Comparison of social-ecological system state variables. The UK NEA 2011 variables originate at a zero value; the WP 4 variables describe a balance between two extremes.

	UK NEA 2011 (Haines-Young *et al.*, 2011)	**WP 4 (this report)**
1	*Environmental awareness* describes the level of appreciation and concern for conservation and sustainability issues in society, for example recycling;	The *value-orientation axis* specifies whether people mainly act as consumers, seeking satisfaction of their own, or their families', 'well-being needs', or instead whether they act collectively, as members of a community. This in turn determines the comparative importance of, on the one hand, the market, and on the other hand, 'collective arrangements'.
2	*Governance and intervention* describes how much the state uses political authority and institutional resources to manage society;	The *governance axis* describes the distribution of legitimate power in society. 'Interdependence' refers to dominance by large-scale governance (sovereign states, continental federations, or the world as a whole). 'Autonomy' refers to the extent that power is devolved to the smallest institutional scale.
3	*Human well-being* relates to the standards of health provision, education, employment, freedom, human rights and happiness;	
4	*Overseas ecological footprint* is a measure of demand on the earth's resources overseas (resulting from imports of biomass and energy and exports of waste products);	
5	*Adaptation capacity* relates to societies' ability and willingness to cope with the impacts if climate change	

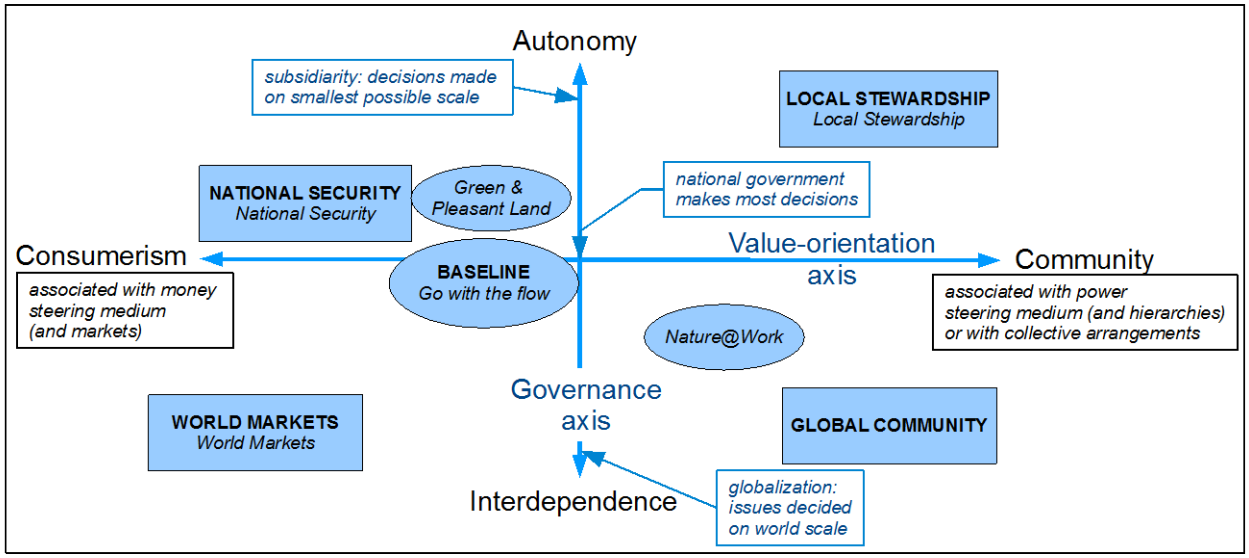


Figure 4.23. The state-space of Figure 4.14 updated, with axes commented and all NEA2011 (Haines-Young *et al.* 2011) scenarios included. The placement of the 'Green & Pleasant Land' and 'Nature@Work' scenarios is indicative rather than definitive, because Haines-Young *et al.* described their scenarios with five dimensions rather than our two. The value axis is labelled in terms of societal arrangements for resource allocation and in terms of Habermas (1984) 'steering media'. It should also be understood in terms of individual values.

4.3.5 Conclusions

Despite the reservations expressed above, we conclude that expert opinion (i) is moderately sanguine about the future for marine ecosystem services, given the continued implementation of existing and foreseen environmental protection legislation, but (ii) considers that the present situation (i.e. the Baseline scenario) is suboptimal: improvements in services could be obtained by drawing on options available under other scenarios, as exemplified in **Section 4.3.4.2**.

The ecosystem changes sketched out in the scenarios now have to be sufficiently quantified to allow the construction of indicators of change, which then become part of an on-going monitoring process.

4.4 Identification of a practicable set of ecosystem indicators for coastal and marine ecosystem services

4.4.1 Introduction

Gibbs (2012) suggests that 'in order to manage something as complex as a marine ecosystem, indicators are needed that are able to provide an insight into the behaviour, state and trajectory of the system'. An indicator can be described as 'a measure or metric based on verifiable data that conveys information about more than just itself' (UNEP-WCMC, 2009). They can be of two types – firstly, specifically ecosystem indicators are 'measures of key ecosystem properties reflecting changes in ecosystem services and can provide information on the direction and possible magnitude of the impact or response of an ecosystem to stress' (van den Belt & Costanza, 2011). Secondly an indicator can be a quantitative value against which change is measured and where the value to be exceeded is incorporated in a statutory or policy instrument, where compliance with it is judged by monitoring (McLusky & Elliott, 2004). Indicators can reflect state and/or performance of the marine system, and they can also reflect the marine ecosystem natural capital stocks and the flow of ecosystem services of significant value (benefits) to human society.

Smeets and Weterings (1999) present the European Environment Agency typology with indicators classified into four types which address the following questions: '*What is happening to the environment and to humans*?' (Type A or descriptive indicators); '*Does it matter*?' (Type B or performance indicators); '*Are we improving*?' (Type C or efficiency indicators), and '*Are we on the whole better off*?' (Type D or total welfare indicators). These may include red flag/tipping point/threshold indicators and early warning indicators which can be an aid to management.

Aubry and Elliott (2006) suggest environmental indicators should have three basic functions:

- *to simplify:* amongst the diverse components of an ecosystem, a few indicators are needed according to their perceived relevance for characterising the overall state of the ecosystem;
- *to quantify:* the indicator is compared with reference values considered to be characteristic of either 'pristine' or heavily impacted ecosystems to determine changes from reference or expected conditions; and
- *to communicate:* with stakeholders and policy makers, by promoting information exchange and comparison of spatial and temporal patterns.

The first of these recognises that a key challenge in the development and use of ecological indicators is to find 'which of the numerous measures of ecological systems characterise the entire system yet are simple enough to be effectively and efficiently monitored and modelled' (Dale & Beyeler, 2001). Indicator choice needs then to be grounded within a conceptual framework such that their individual indicator characteristics are not overemphasised as formal selection criteria and greater attention is given to the function of the indicator within an analytical problem solving logic. All indicators selected should have a function recognising the inter-relations and causality within the environmental system (Niemeijer & de Groot, 2008).

Further, on the second function above, it is axiomatic that management of the marine system requires measurement and that the latter implies monitoring to provide those measurements (Elliot, 2011). In the detection of change, those monitoring measures have to be against a desired outcome, for example

a baseline, reference condition or trigger or threshold value (Gray & Elliott, 2009) and ideally an action is defined *a priori* before the indicators and monitoring are employed. Each of these measures are then indicators which highlight a deviation from change; for example, the WFD, MSFD, Habitats and Species Directive and Environmental Impact Directive are all based on a knowledge of what an area should be like (its 'normal' condition) and whether it has deviated (or will in the future deviate) from this due to human activities. Hence there is a need for indicators to determine the state of that normal condition and the degree of deviation.

On the third function, communicating the compliance with or deviation from indicators is a further challenge. In the present context, ecosystem service indicators are by their nature inherently interdisciplinary and so finding a language common to all stakeholders is not easy, particularly when combining differing philosophies, paradigms and research techniques. A common language will also be called upon to communicate objectives, methods and outcomes to a number of different audiences, from the lay-person through to specialists and policy makers (UNEP-WCMC, 2011).

The literature also refers to the SMART characteristics of indicators, which follows from the work of Doran (1981). According to this set of criteria, in order to be operational, valuable and successful, the management of the environment requires indicators which are Specific, Measurable, Achievable / Appropriate / Attainable, Realistic / Results focused / Relevant, and Time-bounded / Timely. Without meeting these five criteria, it is suggested that the indicators cannot be used in measuring, monitoring and managing change (e.g. Dauvin *et al.,* 2008; Gray & Elliott, 2009). As an extension of this, Elliott (2011) indicated 18 characteristics of the indicators and the required monitoring (**Box 4.2.**). Hence in linking these to the management framework, SMART indicators are needed and indicators fulfilling those 18 attributes for the *Pressures*, *State Change* and *Impact* elements of the DPSIR framework (Gray & Elliott, 2009).

Box 4.2. Properties required/typology of indicators linked to monitoring. Source: modified from Elliott (2011).

- Anticipatory
- Biologically/environmentally important
- Broadly applicable and integrative over space and time
- Concrete / results focussed
- Giving continuity over time and space
- Cost-effective in monitoring
- Grounded in theory / relevant and appropriate
- Interpretable
- Low redundancy
- Measurable
- Non-destructive
- Realistic / attainable (achievable)
- Responsive feedback to management
- Sensitive to a known stressor or stressors
- Socially relevant
- Specific
- Time-bounded
- Timely

Several studies have attempted to identify indicators for ecosystem services, although these studies do not identify indicators specifically for changes in UK marine ecosystem services. UNEP-WCMC (2011) assessed ecosystem indicators for the various categories of ecosystem services (provisioning, regulating, supporting and cultural) based on evidence from 34 sub-global assessment reports. The study considered a wide range of ecosystem types including coastal, cultivated, dryland, forest, inland water, island, marine, mountain and urban regions. Indicators were identified for 23 ecosystem service categories following the TEEB framework which is generic across ecosystems. A distinction was made between state indicators (how much of the service is present) and performance indicators (how much can be used/provided in a sustainable way) following de Groot *et al.* (2010a; 2010b). The study suggested that indicators of ecosystem services were underdeveloped and failed to convey a complete picture, with the average quality of ecosystem service indicators and data availability being considered poor, particularly as a decision support tool (indicators of cultural, supporting and regulating services being particularly limited).

Kandziora *et al.* (2012) proposed a set of ecological indicators, based on an adapted list of ecosystem services from the Millennium Ecosystem Assessments (MEA, 2005), to show the interrelations between ecosystem properties, biodiversity, ecosystem integrity, ecosystem services and human welfare. Their study concluded that ecosystem service indicators meet the criteria of being adequate human–environmental system indicators and therefore, they are an appropriate instrument for decision making and management.

Liquete *et al.* (2013) use a meta-analysis to systematically review the current status and future prospects for the assessment of marine and coastal ecosystem services. They identify 145 papers which specifically assess marine and coastal ecosystem services, mainly with a focus on mangroves and coastal wetlands in Europe and North America. A catalogue of 476 ecosystem service indicators was created and identified gaps in current knowledge. Most indicators relate to a limited set of ecosystem services and benefits, including food provision (fisheries), water purification, coastal protection and recreation/tourism. The findings of this systematic review provide a basis for the planning and integration of future assessments of the provision of marine and coastal ecosystem services.

Böhnke-Henrichs *et al.* (2013) provide a set of ecosystem service indicators within the TEEB (The Economics of Ecosystems and Biodiversity) framework. They argue that the ecosystem service concept has rarely been applied to marine planning and management to date due to a lack of a well-structured, systematic classification and assessment of marine ecosystem services. Their study proposes such a typology and provides guidance on the selection of appropriate indicators in relation to marine spatial planning and management.

Further studies focus solely on the terrestrial environment. For example, Dobbs *et al.* (2011) present a framework for developing indicators for ecosystem services associated with urban forests, using field data, an urban forest functional model, and existing literature. Koschke *et al.* (2012) present a MCA framework for the qualitative estimation of ecosystem services, with benefit transfer and an expert based assessment used to assign values to indicators of ecosystem services to support terrestrial landscape planning.

Following this introduction, **Section 4.4** uses the UK NEA ecosystem services framework which has been described in **Section 4.2**, as an overarching structure to identify a practical set of indicators for components and processes, intermediate services, final services and goods/benefits for the UK marine environment. Examples of national-level data sources available to support indicators for the UK marine

environment are identified. The application of these indicators is examined specifically for six ecosystem services (fisheries and aquaculture, sea defence, prevention of coastal erosion, carbon sequestration/storage, tourism and nature watching, and education) and two broader concepts (biodiversity and cultural assets). In addition, two case studies are presented which demonstrate the importance of site-specific data sources in relation to marine protected areas and managed realignment sites. A discussion of the set of indicators identified by the relevant Descriptor Task teams for assessing GEnS under the MSFD is provided in **Appendix 4.3**.

4.4.2 Ecosystem indicators for coastal and marine ecosystem services

The ecosystem services framework, developed for the UK NEA (2011), recognises the importance of distinguishing between basic processes, intermediate services and final services, and goods/societal benefits (see **Section 4.2.**). Given that the framework was designed to be generally applicable across ecosystems, in order to increase its relevance to the marine environment, Turner *et al.* (2013) modified the framework under the NERC-funded Valuing Nature Network Coastal Management project, and the framework was modified further following UK NEAFO WP 4 (Marine Economics) workshops (see **Figure 4.4.)**. To capture the diversity and complexity of the marine system the indicators need to be specific not only to ecosystem services but also relate to the components and processes and goods/benefits as identified within the framework.

Thus, starting with the framework, a range of ecosystem service indicators have been identified for each category and these were reviewed for their ability to provide insight into the behaviour, state and trajectory of the marine system by the UK NEAFO WP 4 project group at two further workshops. The final lists of ecosystem indicators are presented below (**Tables 4.4.-4.7.**). The indicators contained within the tables are examples and the list is not meant to be exhaustive. These indicators reflect state and/or performance within the marine system and in the case of performance indicators will require a set of associated targets to be established. Some indicators have strong links to management (e.g. the quantity and quality of fish and shellfish, amount of carbon sequestered), while in the case of others it can be argued that these have important interrelationships with ecosystem services (e.g. depth (m); volume (m^3); area of surface (ha); tidal range (m)). For example, water depth (m) as an indicator has relevance for and may impact in various ways on recreational activities, sea defence, coastal erosion, fisheries and aquaculture, and others, but the nature of its relevance and impact is also dependent on other marine components, processes and services. All of the indicators identified are expressed in natural science units or units with more anthropocentric relevance; indicators measured in monetary units are discussed in **Section 4.6** on valuation.

Table 4.4. Indicators of marine ecosystem components and processes and examples of UK data sources.

Marine ecosystem		Indicators (examples of units)	Examples of UK Data Sources
Components, e.g.	**Habitats and species**	Abundance (no.); biomass (g, kg); species diversity (Shannon Wiener Index); % cover of habitat; area of habitat (ha); gene pool; biotope matrix; AMBI (marine biotic index); phytoplankton index.	MESH Atlantic (2004-2008, 2010-2014); UK SeaMap (2006, 2010); European Marine Ecosystem Observatory; Defra MB0102 data layers; OBIS SEAMAP.
	Sea space	Area of surface (ha); volume (m^3); tidal range (m); depth (m); bathymetry; topography.	UK SeaMap (2006, 2010); Scotland's Marine Atlas; Scotland's Marine Plan Interactive; SeaZone Solutions.
	Sea water	Depth (m); volume (m^3); pH; salinity; turbidity (mg/l).	UK SeaMap (2006, 2010); Defra DEM bathymetry maps; Scotland's Marine Atlas; Scotland's Marine Plan Interactive; SeaZone Solutions.
	Substratum	Area (ha) and depth (m) by type (mud, sand, gravel, etc.).	UK SeaMap (2006, 2010); MESH Atlantic (2004-2008, 2010-2014).
Processes, e.g.	**Production**	Community production (kcal); Net productivity by species (kcal/ha/yr); P:B (productivity:biomass) ratios.	EU MyOcean project; Published and grey literature.
	Decomposition	Amount and number of decomposers (n/ha); Decomposition rate (kg/ha/yr).	Published and grey literature.
	Food web dynamics	Changes over time in community composition (abundance (no.); biomass (g, kg); species diversity (diversity indices)); population dynamics (age classes, male:female ratios).	DASSH website; Fish trawl surveys database (ICES, 1989 to present); Published and grey literature.
	Ecological interactions (inter- and intraspecific)	Competition for food and space; resilience and resistance (predator:prey, adults:juveniles, etc.)	Published and grey literature.
	Hydrological processes	Current speed (m/s) and direction; wave height; changes in temperature (°C); changes in salinity; changes in turbidity (mg/l); NAO (North-Atlantic Oscillation) cycles.	Defra MB0102 data layers; UK SeaMap (2006, 2010); EU Global Ocean OSTIA Sea Surface Temperature and Sea Ice analysis REPROCESSED (1985-2007).
	Geological processes	Sediment accumulation rates; slopes; seabed form; channel depths; erosion-deposition cycles.	UK SeaMap (2006, 2010).
	Evolutionary process	Changes in genetic diversity; mutation rates; influx/efflux of species (no.).	Published and grey literature.

Table 4.5. Indicators of intermediate ecosystem services and examples of UK data sources.

Intermediate ecosystem services		Indicators (examples of units)	Examples of UK Data Sources
Supporting services	**Primary production**	Quantity of primary production (g C per unit area/volume); Quality of primary production (e.g. efficiency of converting sunlight to carbon).	Published and grey literature.
	Larval and gamete supply	Quantity of larvae/gametes supplied to a particular location (number per m^3); Quality of larvae/gametes supplied to a particular location (% affected by disease; mortality rates); link to hydrological processes.	UK spawning and nursery grounds (Ellis *et al.*, 2012); Fish eggs and larvae database (ICES, 1967-present).
	Nutrient cycling	Changes (output of the system less input to the system) in the amount of nitrates, phosphates, silica (g per unit area/volume); Denitrification (kg N/ha/yr).	Published and grey literature; Cefas modeling data.
	Water cycling	Changes (output of the system less input to the system) in the amount of water (m^3).	Relevant Environment Agencies (EA, SEPA, NIEA); Published and grey literature.
	Formation of species habitat	Change in area of habitat (per ha); change in quality of habitat; change in number of juveniles; deviation of hydrographic processes.	MESH Atlantic (2004-2008, 2010-2014); UK SeaMap (2006-2010).
	Formation of physical barriers	Change in amount of natural barriers e.g. salt marsh, reefs, sand dunes, reed beds etc (% cover, ha).	MESH Atlantic (2004-2008, 2010-2014); UK SeaMap (2006-2010).
	Formation of seascape	Changes in area by scenic type (ha, % cover, visual range (m, km)).	MESH Atlantic (2004-2008, 2010-2014); UK SeaMap (2006-2010).
Regulating services	**Biological control**	Quantity of pest/disease/predator-control species (number); Quality of pest-control species (prevalence).	DASSH website; Published Literature.
	Natural hazard regulation	Width or area (and volume if applicable) of salt marsh, reed bed, mudflat, sand dunes etc. (m, % cover, ha, m^3) absorbing energy.	MESH Atlantic (2004-2008, 2010-2014); UK SeaMap (2006-2010).
	Waste breakdown and detoxification	Water quality indicators (N mg/l, P mg/l, bacterial levels mg/l etc.); total dissolved solids (mg/l); Water volume; Assimilative capacity.	Relevant Environment Agencies (EA, SEPA, NIEA).
	Carbon sequestration	Amount of carbon sequestered (tonnes of CO_2 per m^2 or m^3); assimilative and recycling capacity, net carbon burial (tonnes per hectare per year).	DASSH website.

Table 4.6. Indicators of final ecosystem services and examples of UK data sources.

Final ecosystem services		Indicators (examples of units)	Examples of UK Data Sources
Provisioning services	**Fish and shellfish**	Fish and shellfish population size (biomass of fish/shellfish in tonnes); Quality of the fish, shellfish (age profile; length profile; % affected by disease; mortality rates).	UK spawning and nursery grounds (Ellis *et al.* 2012); National Fish Population Dataset (EA, 2004-2014); Fish trawl surveys database (ICES, 1989 to present); DASSH website.
	Algae and seaweed	Quantity of seaweed stock (biomass in tonnes, area of seaweed ha); Quality of seaweed stock (% affected by disease; mortality rates).	DASSH website.
	Ornamental materials	Quantity of raw material (tonnes); Quality of raw material (concentration).	Published and grey literature.
	Genetic resources	Quantity of species with potential/actual useful genetic raw material (tonnes); Gene bank composition (e.g. number of species and subspecies); Quality of species with potential/actual useful genetic raw material (tonnes equivalent if variation in quality).	Published and grey literature.
	Water supply	Quantity of water extracted for (e.g.) irrigation, cooling and ballast.	Relevant Environment Agencies (EA, SEPA, NIEA); Charting Progress 2.
Regulating services	**Climate regulation**	Greenhouse gas balance especially carbon sequestration (g C); Quantity of greenhouse gases fixed and/or emitted; Effect on climate parameters (temperature, rainfall, wind, etc).	Published and grey literature.
	Natural hazard protection	Width or area of salt marsh, reed bed, mudflat, sand dunes etc. providing natural hazard protection (m, % cover, ha); sediment stabilisation properties; water retention capacity (m^3); (wave) energy dissipation capacity.	UK SeaMap (2006-2010); MESH Atlantic (2004-2008, 2010-2014).
	Clean water and sediments	Amount of waste that can be recycled or immobilised (tonnes); Biological oxygen demand (mg O_2/litre/day); Amount of organic matter in water and sediment (mg/l); Amount of heavy metals in water and sediment (mg/l); Amount of bacteria in water and sediments (mg/l); Heavy metal (and other pollutant) content in marine organisms (concentration).	Relevant Environment Agencies (EA, SEPA, NIEA).
Cultur	**Places and seascapes**	Number of designated sites; Number/area of specific seascape features; % of total natural seascape.	JNCC website; English Heritage; National Trust.

Table 4.7. Indicators of goods/benefits and examples of UK data sources.

Goods/benefits		Indicators (examples of units)	Examples of UK Data Sources
Provisioning services	**Food (wild, farmed)**	Nutrition from seafood consumption (g protein/year or g protein/year/head or per household); Fish landed for human consumption (landings data at particular times and places in tonnes).	MMO Landings data; FAO Statistics; Office for National Statistics.
	Fish feed (wild, farmed, bait)	Nutrition from non-human seafood consumption (g protein/year); Fish landed not for human consumption (landings data at particular times and places in tonnes); Bait landed for angling (tonnes); Quantity of bait collected by type.	MMO Landings data; FAO Statistics; Office for National Statistics.
	Fertiliser and biofuels	Mineral and other content used (e.g. N concentration in g, tonnes); Quantity of biomass harvested for energy production.	Published and grey literature.
	Ornaments and aquaria	Ornamental use (tonnes) by type; Number of people/businesses who rely on ornamental artefacts (no.).	Published and grey literature.
	Medicines and blue biotechnology	Contribution to medicines (number of medicines, improvements in mortality rates and quality of life, etc.); Total amount of useful substances that can be extracted (kg/ha); Quantity of specific blue biotechnologies (e.g. biocatalysts).	Published and grey literature.
Regulating services	**Healthy climate**	Physical damage avoided through net GHG sequestration and effects on climate parameters; bodily harm avoided (lives saved and injuries not incurred) through net GHG sequestration and effects on climate parameters.	Published and grey literature.
	Prevention of coastal erosion	Number of prevented hazards (no./yr); avoided displacement of residents/businesses (number of people, m^2 of buildings); quantity of risk prevention (quantity of assets affected adjusted for risk).	Local Authorities; National Coastal Erosion Risk Mapping (NCERM) data.
	Sea defence	Amount of man-made infrastructure no longer required; Businesses and people protected from flooding; Number of flood related mortalities; Flooding days per year (combined with rainfall indicator).	Relevant Environment Agencies (EA, SEPA, NIEA); Local Authorities; Environmental Valuation Reference Inventory (EVRI).
	Waste burial / removal / neutralisation	Quantity of degradable waste deposited (tonnes by type); Quantity of non-degradable waste deposited (tonnes by type); Pollution damage avoided by not disposing degradable and non-degradable	Relevant Environment Agencies (EA, SEPA, NIEA); Cefas; Local Authorities; Industry discharge records; Water Companies.

Goods/benefits		Indicators (examples of units)	Examples of UK Data Sources
		waste elsewhere (type and extent); Treatment and engineering works not required (type and capacity); Changes in activity not implemented due to capacity to immobilise waste (quantity and/or other characteristics of activity).	
Cultural services	**Tourism and nature watching**	Number of participants (no./yr); Number of facilities (number visitors per facility/yr); Amount of time spent participating (hours/days).	Office for National Statistics; UK Centre for Economic & Environmental Development (CEED); Great Britain Tourism Survey; OBIS SEAMAP; RSPB statistics; Royal Yachting Association.
	Spiritual and cultural well-being	Sites with cultural heritage/wellbeing (usage rates by people, degree of importance); Sites with spiritual and/or religious significance/wellbeing (number of people who attach significance, degree of significance attached).	Office for National Statistics; Economic and Social Data Service (ESDS).
	Aesthetic benefits	Number and/or area of marine features of given stated appreciation; Length of Heritage Coast (km).	Office for National Statistics; Economic and Social Data Service (ESDS).
	Education	Field trips (number and number of people involved); Classes (numbers and number of people involved); Scientific studies (number of research papers, subscriptions, library borrowing, on-line downloads); Books (number, print run, library usage, e-book downloads); other publications including newspaper articles (circulation including on-line accessing); works of art (number of works, number of people viewing work).	Office for National Statistics; UK Directory of the Marine Observing Systems (UKDMOS); School and University Reports; Charting Progress 2.

Examples of UK data sources have been identified for each category of ecosystem service indicator (**Tables 4.4.-4.7.**). These sources provide good spatial coverage at the UK-level, and contain both observed and modeled data. However, the data tend to vary in their spatial accounting unit and the frequency of reporting. This list of sources is not exhaustive, but is used to show the range of national sources currently available. Where UK-level data sets have not been identified, evidence may be available from the published and/or grey literature.

Although beyond the scope of this report, there may be a need for a further set of indicators which show the emergent properties of ecosystems. The emergent properties are linked to the fundamental processes of the system which provide the conditions for the delivery of ecosystem services. These properties include resistance, as the ability of a system to withstand the pressure caused by a potential stressor before it changes, and also resilience which is defined as the ability of the system to recover after the addition of a stressor (see Elliott *et al.*, 2007). Indicators may relate to the structure of the system, i.e. attributes at one time, or functioning related to rate processes. Indicators measuring the vulnerability of the system would allow policy makers to prioritise areas for action (e.g. Pethick & Crooks, 2000). Similarly, indicators may be required of the ecological or socio-economic carrying capacity of the system and its ability to support ecological components or the human activities present (see Elliott *et al.*, 2007 and references therein).

Based on the list of indicators presented above, further consideration is given to six ecosystem services comprising fisheries and aquaculture, sea defense, prevention of coastal erosion, carbon sequestration/storage, tourism and nature watching, and education, and two broader concepts comprising biodiversity and cultural assets. These were selected for consistency with the review of economic valuation studies (**Section 4.6**) and are used to demonstrate how multiple indicators may be necessary to capture the complexity of the marine system associated with even single ecosystem services and to detect change over time in their service provision. However, it is unlikely that indicators of all elements of the framework will be used in such a case hence we identify a sub-set of the framework.

Figure 4.24. focuses on ecosystem indicators associated with **fisheries and aquaculture**, and specifically relates to the supply of food for human consumption as the good/benefit. While the final ecosystem indicators are necessarily species-specific, it is recognised that the intermediate services linked to the two final ecosystem services 'fish and shellfish' and 'algae and seaweed' are likely to be broadly similar and no single indicator is likely to satisfy all requirements operationally. In the case of fisheries and aquaculture much of the relevant complexity of the marine system can be captured by the first three columns of the framework without explicit reference to indicators of the marine components and processes. However, quantification of the relevant indicators may be difficult but these indicators are important for effective intervention for example when interest is in the delivery of catches of certain species. This example also exemplifies the openness of the marine system compared to the terrestrial and freshwater systems in that the delivery of an ecosystem service at one area, and thus the compliance with an indicator, may not be dependent on areas outside that being considered; in essence the size of the fishable stock at one area around the UK coast may depend on feeding and nursery grounds elsewhere, especially adjacent estuaries. Similarly, the ability of an area to support bird populations, important for nature watching, depends on their feeding and breeding grounds possibly in different latitudes (Elliott & Whitfield, 2011). The interpretation of such good/benefit indicators would need to be supported by knowledge of the complementary capital employed and the importance of factors such as migrations.

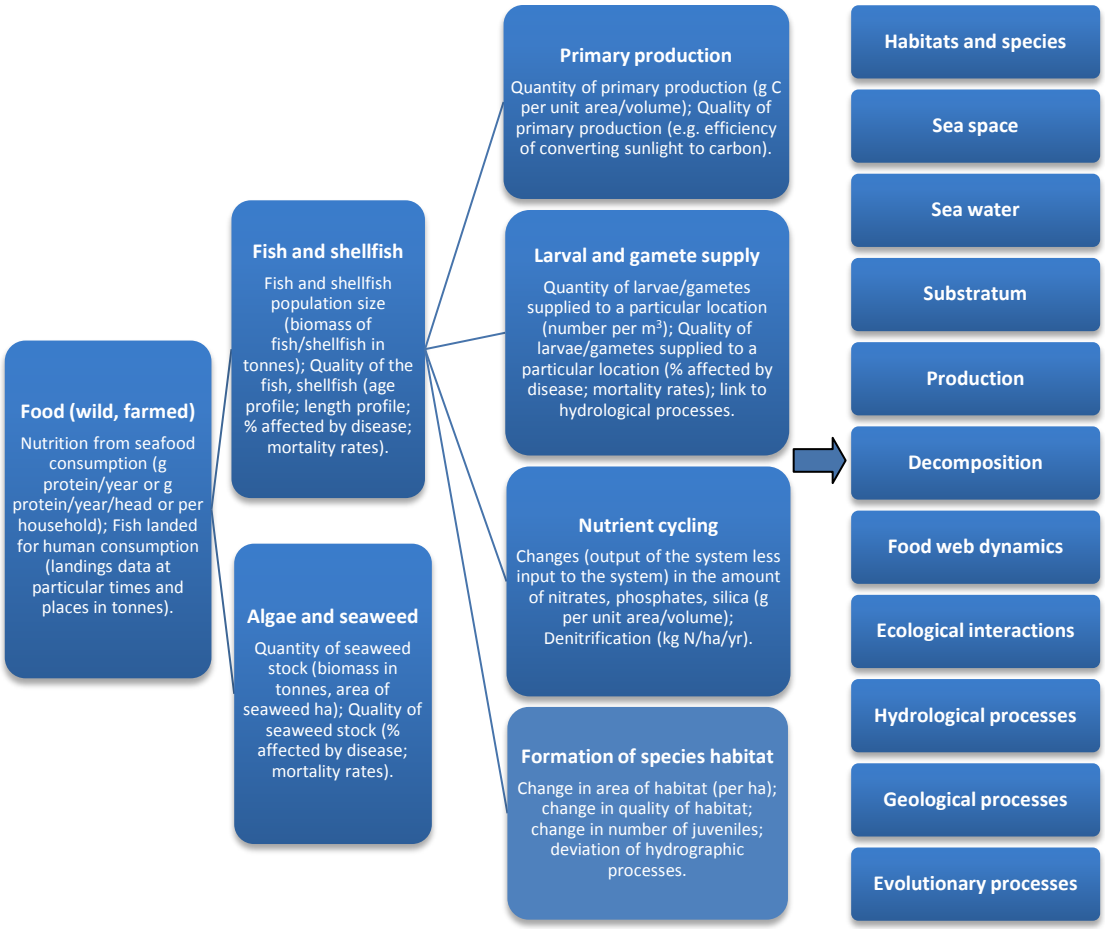


Figure 4.24. Fisheries and aquaculture.

The **sea defence** good/benefit directly links to natural hazard protection, a final ecosystem service, and natural hazard regulation, an intermediate ecosystem service (**Figure 4.25.**). At least six ecosystem components and processes can be identified in the framework that can cause *State changes* that lead to *Impacts* on sea defence provision. Depending on the context, indicators drawn from the the goods/benefits, final and intermediate ecosystem services, may have relevance.

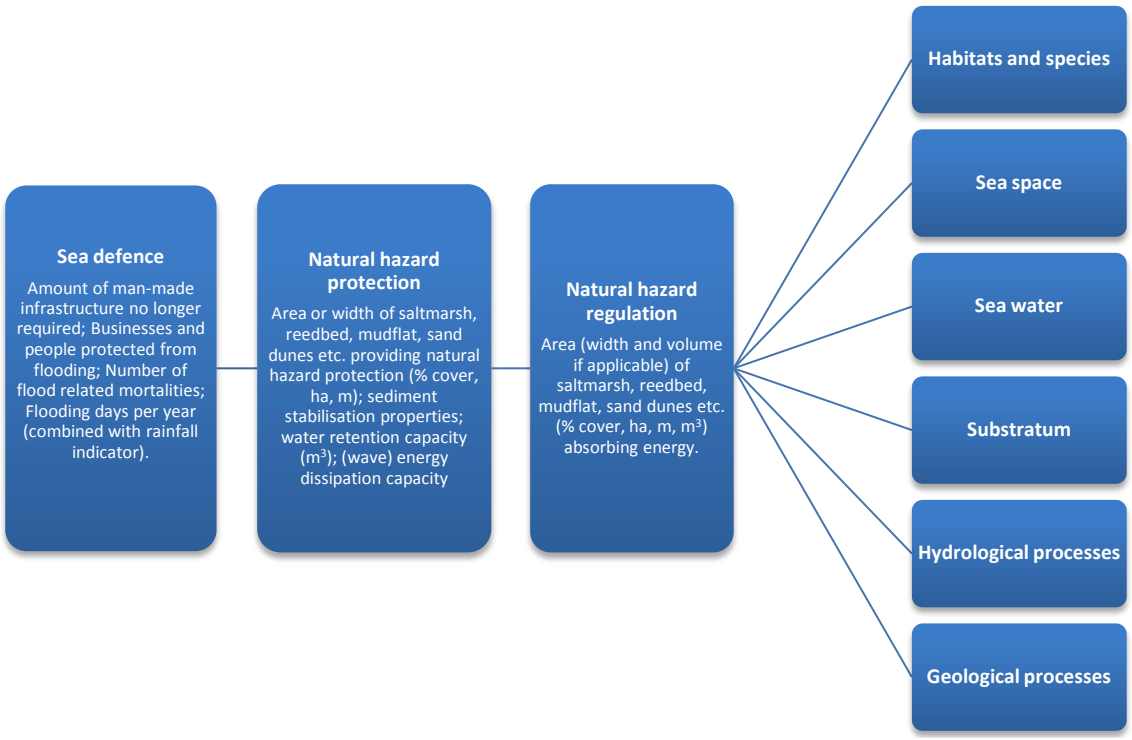


Figure 4.25. Sea defence.

In the case of **prevention of coastal erosion**, once again indicators can be drawn especially from the goods/benefits, final and intermediate ecosystem services with the selection of indicators being dependent on the context (**Figure 4.26.**).

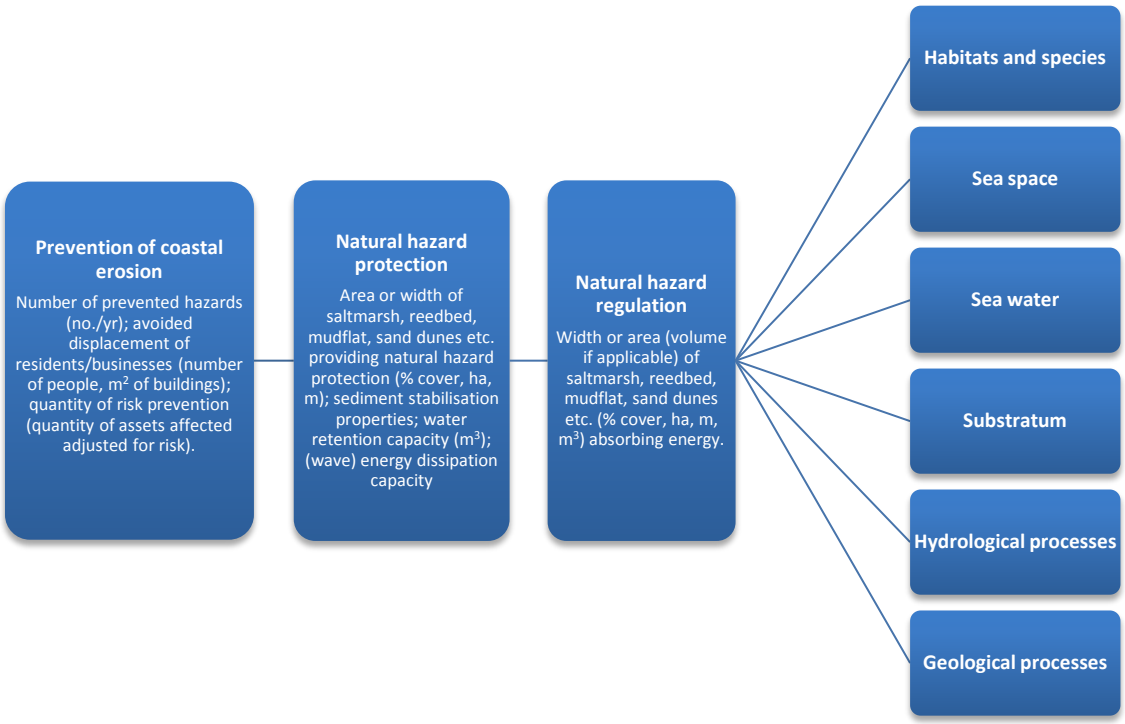


Figure 4.26. Prevention of coastal erosion.

With regard to **carbon sequestration/storage**, identified here as an intermediate ecosystem service, it may be sufficient to use indicators at the levels of marine components/processes, intermediate ecosystem services level (in the form of indicators of climate regulation), and/or indicators of a healthy climate (goods/benefits) to reflect the complexity of the system (**Figure 4.27.**). Focusing on the marine components and processes, multiple indicators may be required as, for example, both biological indicators (relating to species and habitats) and physical indicators (relating to substratum, geological processes, etc) create capacity for carbon sequestration with the outcome dependent on the ecological and physico-chemical characteristics of the specific site. The complexity of the coastal and marine sites combined with availability of data may dictate the need to focus on indicators of 'climate regulation' or of 'healthy climate' as listed in **Figure 4.27.**

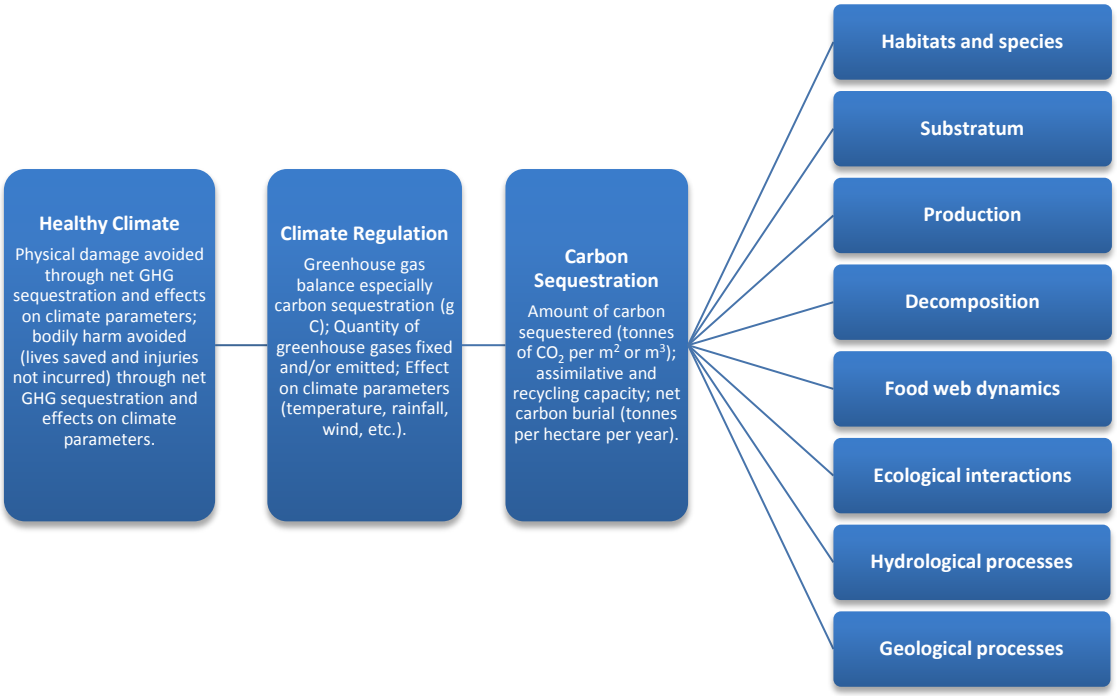


Figure 4.27. Carbon sequestration/storage.

In relation to **tourism and nature watching,** five key final services have been identified as being most relevant and include the provisioning services of 'fish and shellfish', 'algae and seaweed' and 'ornamental materials', the regulating service of 'clean water and sediments', and the cultural service of 'places and seascape' (**Figure 4.28.**). Although not captured within **Figure 4.28.**, indicators at the intermediate and marine components and processes level will also be important as these set up the fundamental system leading to the provision of tourism and nature watching. For example, indicators of fish abundance (marine component) may be important for recreational angling activities, indicators of sea water turbidity or depth (marine component) may be important for bathing, indicators of hydrological (marine) processes for sailing, and indicators of the formation of specific habitat (intermediate service) for recreational diving.

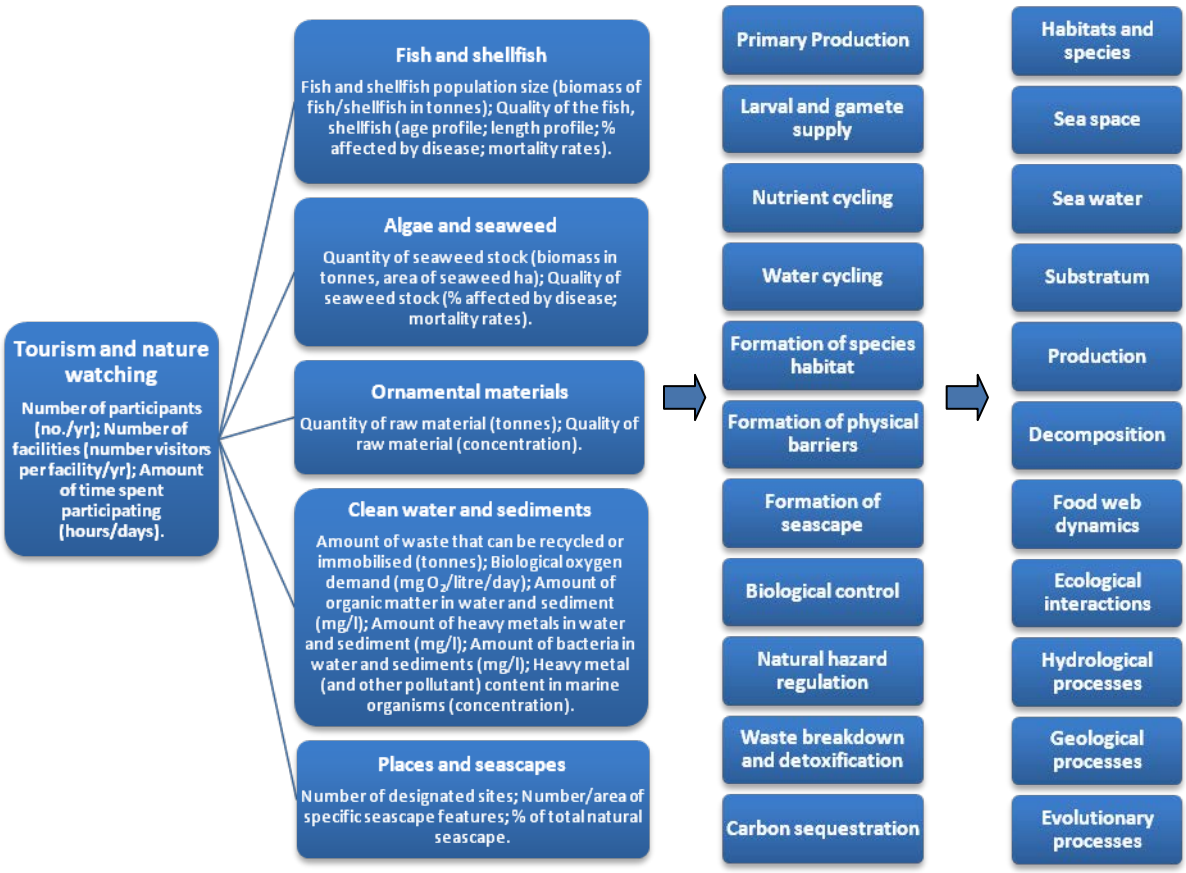


Figure 4.28. Tourism and nature watching.

Several indicators are highlighted for **education** as a good/benefit (**Figure 4.29.**) although for education the focus is likely to be on marine components or processes (e.g. habitats and species), intermediate services (e.g. nutrient cycling) and/or final services (e.g. genetic resources) and, therefore, indicators would have to be issue-specific. The relationship between education and such indicators may not be straight forward, for example indicators of habitats and species and education may be related, but arguably both positively and negatively correlated as both improving and declining habitats and species provide opportunities and stimulus for research. A wider perspective on education may also be obtained using indicators of other goods/benefits which are not depicted in **Figure 4.29.**, for example education relating to fish landed for human consumption.

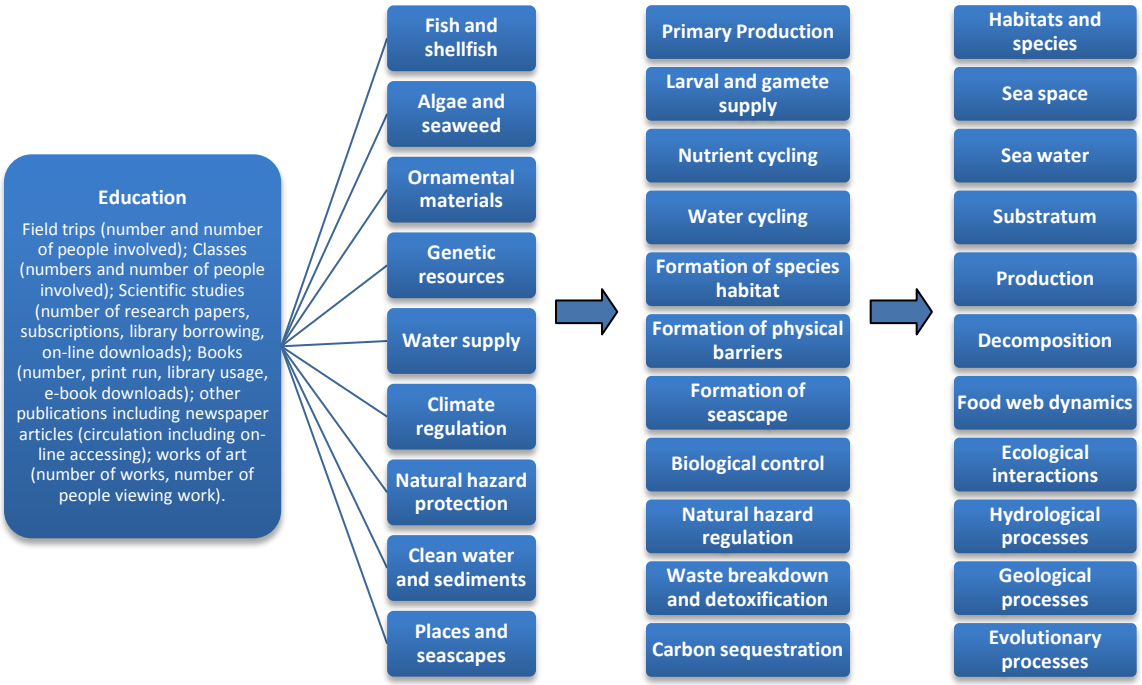


Figure 4.29. Education.

With regard to **biodiversity** or **cultural assets**, it is unlikely that indicators from one element of the ecosystem services framework alone will suffice. For example, applying the ecosystem services framework given in **Figure 4.4**., it is clear that **marine biodiversity,** a site-specific attribute which underpins the provision of marine ecosystem services, would be most appropriately captured using a range of indicators associated with those marine components of habitats and species, recognising that these indicators reflect properties of a complex marine environment (**Figure 4.30.**). Similarly for **cultural assets**, this might be captured by indicators for a range of final ecosystem services, such as the provision of 'fish and shellfish', 'clean water and sediments', and 'places and seascapes', and to a range of goods/benefits including provision of 'sea defence', proximity to 'tourism and nature watching' opportunities, 'spiritual and cultural well-being', and 'aesthetic benefits' (**Figure 4.31.**).

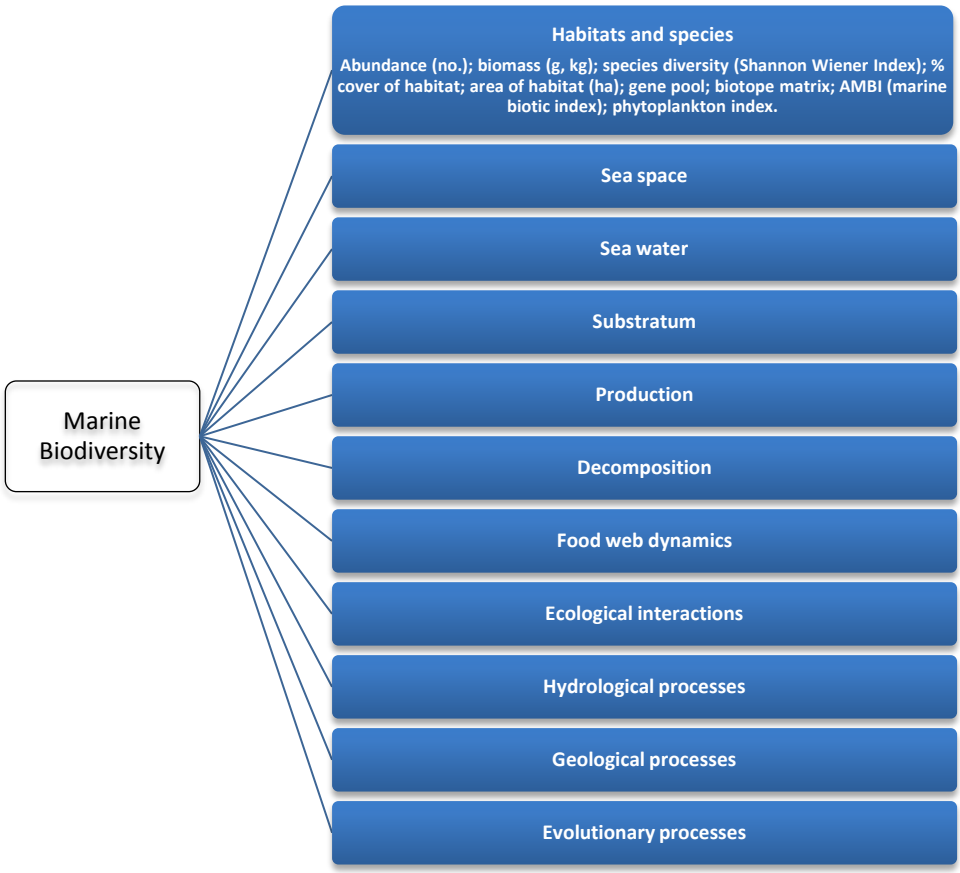


Figure 4.30. Marine biodiversity.

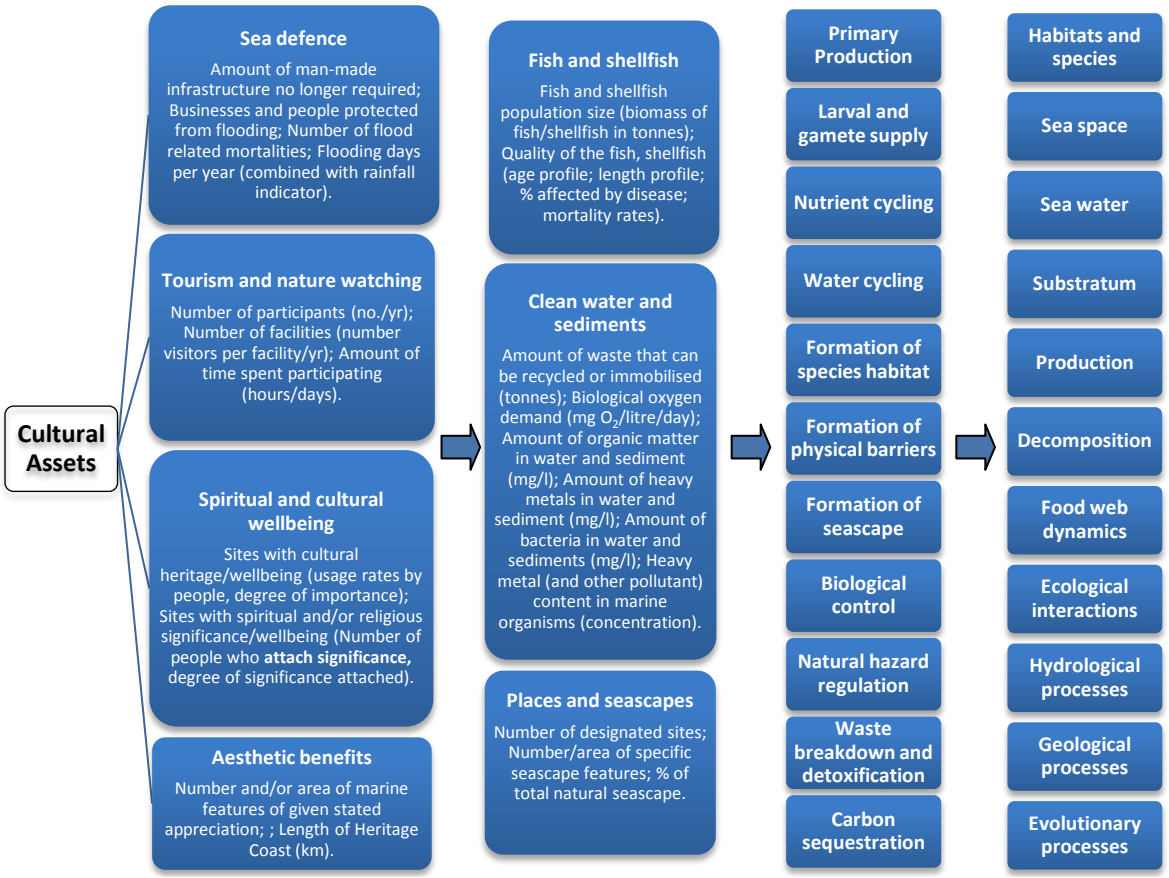
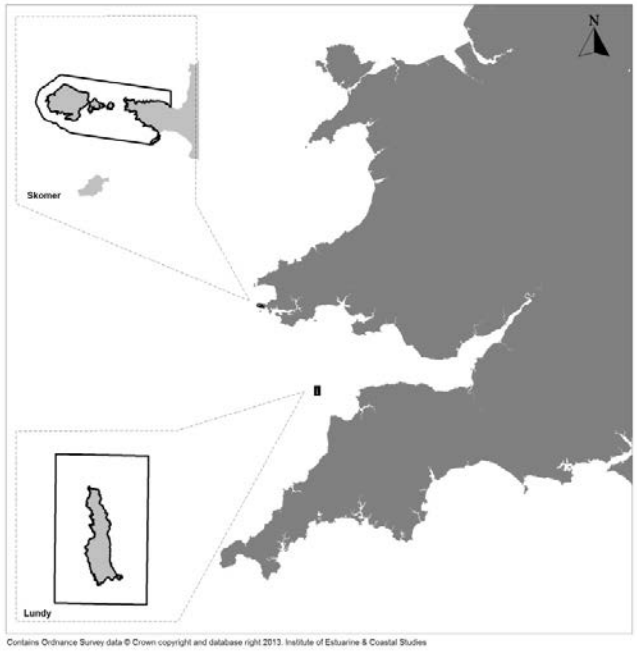


Figure 4.31. Cultural assets.

4.4.3 Case Studies

4.4.3.1 Marine Protected Areas

In the context of Marine Protected Areas (MPAs), Potts *et al.* (2013) recognise the importance of ecosystem service provision by existing and proposed UK MPAs, for example the Lundy No Take Zone (NTZ) and Skomer Marine Nature Reserve (MNR) (**Figure 4.32.**). The requirement for a suite of ecosystem service indicators to support the selection, monitoring and evaluation of any associated marine management measures is apparent. The data for such indicators are likely to be drawn from site-specific published and unpublished sources given the finer resolution which would be required at a more localised level. Some examples of the site-specific data sources are provided by a partial review of evidence relevant to ecosystem service indicators for the two MPA cases.



European Lobster (*Homarus gammarus*)

Juvenile Black-legged Kittiwake (*Rissa tridactyla*)

Figure 4.32. Location of Lundy and Skomer MPAs and images of local wildlife. Source: Institute of Estuarine & Coastal Studies (IECS), University of Hull.

The Lundy NTZ was designated in 2003 in order to protect marine wildlife while improving local fish stocks, and is located to the east of Lundy Island, within the wider Lundy Marine Conservation Zone (MCZ). Evidence for the site can be found in annual Lundy Field Society reports (dating back to 1947) and in more recent studies which focus specifically on the impacts of the NTZ (Hoskin *et al.*, 2009; 2011; Wootton *et al.*, 2012; Coleman *et al.*, 2013). Evidence suggests that there have been changes to the provision of a number of ecosystem services since its designation, and these have included improvements in local shellfish stocks (a final ecosystem service), potential spill-over effects to the local fishery (a good/benefit), and improvements to local tourism/nature watching (a good/benefit) reflected in on-site recreational diving. Examples of the available ecosystem service indicator evidence include:

- Hoskin *et al.* (2009) on the first five years of the NTZ suggest that for local shellfish stocks there is a change in the size profile of the population with a 5% increase in the size of European lobster (*Homarus gammarus*) and a 427% increase in abundance of European lobster within the Lundy NTZ. It is suggested that an observed 97% increase in the abundance of undersized lobsters within the NTZ and 124% increase in its abundance in waters adjacent to the NTZ boundary provides potential evidence of a spill-over effect to the local lobster fishery – an assessment from this extended area of lobster landings for human consumption would be required to confirm this suggestion (refer to **Figure 4.24.**). These findings are supported by Wootton *et al.* (2012) who used similar methods to demonstrate positive effects of the Lundy NTZ on increased lobster abundance and size within the NTZ.
- Wootton *et al.* (2012) demonstrates the apparent negative effects of the NTZ such as increased injury and shell disease. Their study raises concerns about the impact that greater population densities has on disease outbreaks (% with disease), with evidence suggesting that high severity shell disease in the Lundy NTZ was significantly associated with injury, for example injured male

lobsters within the NTZ were over three times more likely to possess the high severity form of shell disease.

- The wider Lundy MCZ attracts a large number of participants in recreational diving (refer to **Figure 4.28.**); it is estimated 1,370 recreational diver days (1 person diving for 1 day) occur at Lundy each year, around 60% of which occur within the NTZ (equating to 820 diver days) (MCZ Project, 2012). An improvement in the condition of site features, including any associated increase in abundance and diversity of species, could improve the quality of diving at the site.

Skomer MNR, which includes the waters around Skomer Island, Middleholm and parts of the Marloes Peninsula in South Pembrokeshire (**Figure 4.32.**) is currently the only MNR in Wales; it was designated in 1990 previously having been a voluntary MNR, and is managed by Natural Resources Wales. Evidence to support the use of a suite of indicators can be found in the Skomer MNR annual reports (providing some data series back to 1987), various reports produced by the Countryside Council for Wales (now Natural Resources Wales), Joint Nature Conservation Committee, and similar agencies. Examples of such evidence include:

- The CCW reports that the restriction of mobile fishing gears within the Skomer MNR has increased the abundance of the local King scallop (Pecten maximus) population 'at least four fold and perhaps more than eight fold' over the first 20 years of its designation (CCW Press Release, 20 April 2010).
- Taylor *et al.* (2012) report the number of breeding birds (using the recognised method of counting Apparently Occupied Nests, AON) and breeding success of Black-legged Kittiwakes (*Rissa tridactyla*) on Skomer Island between 1989 and 2012. In 2012, the breeding numbers of Black-legged Kittiwake totalled 1,594 AON, which was a 13.23% decrease on 2011 breeding numbers and a 30.15% decline over the last five years (2007-2012). The success of fledging Black-legged Kittiwakes is a recognised OSPAR Ecological Quality Objective (EcoQO) and was reported for 2012 at three sites on Skomer Island (S.Stream, High Cliff, and The Wick), encompassing 37% of the total breeding population. The mean success of fledging Black-legged Kittiwakes was reported as 0.32 per AON for 2012, which is below the 23-year mean of 0.64 per AON since monitoring began.
- Monitoring of Grey seal pups in 2012 indicated that the total pup numbers for the MNR reached 310 which is the highest total ever recorded - pup survival was 76%, which is slightly below the average for the last ten years (Lock *et al.*, 2013).
- The site is also recognised as providing several services associated with tourism/wildlife watching with the number of participants reported annually. For example, for 2012 it was reported that 1,008 diver days (with Lucy wreck located within the MNR a popular dive site), 380 recreational craft visits and 483 anglers (192 shore and 291 boat anglers) were observed within the Skomer MNR (Lock *et al.*, 2013).

4.4.3.2 Managed Realignment

The importance of salt marsh habitat in the UK has been recognised within the literature for the delivery of a number of ecosystem services, including sea defence, prevention of coastal erosion, formation of species habitat for birds, fish and invertebrates, carbon sequestration and tourism/nature watching (Everard, 2009; Fonseca, 2009; Luisetti *et al.*, 2011; Burdon *et al.*, 2011). The restoration of salt marsh habitat, through the implementation of managed realignment (MR) at appropriate sites within the UK may result in the provision of a wide range of ecosystem services. **Figure 4.33.** illustrates wading birds feeding at the Welwick managed realignment site on the Humber Estuary (top image) and its potential to attract nature watching (bottom image). Mander *et al.* (2013) shows the potential for MR sites as wading bird feeding areas but emphasises that while they provide additional feeding time, their prey

carrying capacity (and hence creation of certain ecosystem services) may not be the same as natural areas.

Figure 4.33. Welwick managed realignment site, Humber Estuary. Source: Institute of Estuarine & Coastal Studies (IECS), University of Hull.

In order to assess potential change of ecosystem service provision, a suite of ecosystem service indicators may be required.

- King and Lester (1995) compared the sea defence capacity of man-made sea defence structures with and without a salt marsh buffer. Using the width of salt marsh (in m) as an indicator (refer to **Figure 4.25.**), their study deduced that as the width of vegetation decreases, height of the man-made sea wall would need to increase in an almost linear relationship, with related cost implications. For example, at a site with an 80m width of salt marsh habitat a 3m high sea wall would be required for sea defence provision, whereas if the salt marsh habitat was removed, a 12m high sea wall would be required to provide the same level of sea defence.

- Möller *et al.* (2002) used energy dissipation capacity as an indicator for natural hazard protection (final ecosystem service) which is of relevance to both sea defence and prevention of coastal erosion (refer to **Figure 4.25.** and **Figure 4.26.**), and reported energy dissipation rates of 89% over salt marsh as opposed to 29% over bare sand flats.
- A number of studies have looked at the potential for salt marsh to act as a nursery and/or feeding area for fish species. For example, Fonseca (2009) used the abundance of juvenile sea bass (*Dicentrarchus labrax*) per size class within three MR sites in the Blackwater Estuary, in combination with an estimate of the survival rate to minimum commercial landing size per size class, as an indicator of the potential contribution to local fish stocks. Findings show that the sampled MR sites have the potential to contribute 1.65kg of juvenile bass per hectare of salt marsh (mean value) surviving to minimum landing size (36cm) after 4 or 5 years.
- Salt marsh is recognised as providing an important carbon sequestration service within MR sites (**Figure 4.27.**), for example Luisetti *et al.* (2011) reported net carbon burial values (in tonnes per hectare per year) of 0.266 and 3.347 for sedimentation rates of 1.5mm and 6mm respectively within the Blackwater Estuary.
- MR sites provide potential for recreational activities, with the number of participants per activity being identified as a suitable indicator of the provision of this good/benefit (**Figure 4.28.**). For example, it was reported that the most popular uses of the Paull Holme Strays MR site in the Humber Estuary included walking/running (61% of respondents), enjoyment of the site/fresh air (59% of respondents), dog walking (41% of respondents), bird/nature watching (37% of respondents) and fishing (10% of respondents) (n=117, Environment Agency, 2007b).

4.4.4 Concluding comments

This section has identified a practicable set of ecosystem service indicators, which reflect the State Changes and Impacts within the DPSI(W)R framework. The indicators are grounded within an ecosystem services framework, which recognises the need to distinguish between marine processes and components, intermediate ecosystem services, final ecosystem services and good/benefits. This framework points towards the complexity of the marine system which we try to capture also through the use of indicators. In these ways, the indicators are linked to and support the DSS for adaptive coastal management.

Data requirements are an important consideration in making operational the use of indicators in the marine environment. National data sets exist for indicators of a large number of elements of the framework based on primary observation and modeled evidence for the UK coastal and marine environment. The case studies emphasise the importance of published and/or grey literature when site-specific evidence is sought, where available this tends to be higher resolution data and available for different periods.

Although our focus here has been on ecosystem service indicators relating to the quantification and monitoring of change in provision, the indicators identified could also be applied to test for compliance against a given policy instrument such as Good Environmental Status (GEnS) as required under the MSFD (Borja *et al.*, 2013). The indicators identified here are more specific in nature than those indicators proposed under the MSFD which are currently open to interpretation, in this case by individual Member States, because of the greater variety that characterises the marine environment of the regional seas. In general, we consider that the suite of ecosystem service indicators identified here are consistent but

have a different focus to those proposed for the MSFD. There is a need for indicators of GEnS to incorporate ecosystem services and goods/benefits (Borja *et al.*, 2013).

The DSS for adaptive coastal management will be especially dependent on natural science knowledge about the basic processes and functioning of coastal/marine ecosystems, as well as the dynamics of environmental change. Models provide a way of organising much relevant information in a rational way.

4.5 Modelling coastal and marine environments systems

4.5.1 Introduction

Models offer a way to synthesise our understanding of the environment, analyse changes in ecosystems with complex and non-linear interactions, and forecast future changes. Such models range from conceptual frameworks, through correlation models fitted to data, to models based on rigorous fundamental physical, chemical, biological and ecological theory. Models are inevitably developed based on the conceptual understanding of the system of the community developing the models, and to the extent that such understanding is necessarily incomplete, the models will also necessarily be incomplete.

The complexity of the coastal marine environment represents an intellectual and technical challenge to observational scientists and modellers; incorporating the complexity of human interactions within that environment adds a further layer of complexity. Because of this, even the rigorous models based on fundamental underlying physical theory require some approximations and parameterisations, and as the range of processes and interactions increases, these approximations and parameterisations also increase.

It also must be recognised that models are developed for particular applications. Models can often be adapted to meet other goals, but there is no single model suitable for addressing the wide range of issues of relevance to the UK NEA. Models must therefore be viewed as simplifications and abstractions of the complex environmental reality and as useful tools to help us investigate the systems and consider how it may evolve in the future, but they cannot provide a complete description of complex marine systems. For some purposes very simple models are entirely appropriate to address the research or management questions and for other purposes large complex models are required. It is not a matter that any one type of model is better than the other, but rather which is suitable for the task in question. **Appendix 4** includes a summary of the main issues relating to model reliability and a couple of examples and explanations of terms.

In the next section we offer an overview of the types of marine model tools that are likely to be useful for supporting the UK NEA. This is not intended to be a comprehensive review but does aim to show the breadth of tools that are available and necessary for this task, and also considers if these models are currently able to deliver the UK NEA goals, or are likely to be able to do so in the near future. Since this document is a contribution to the UK NEA, we will draw examples where possible from the UK shelf sea environment (**Figure 4.34.**). A feature of the shelf sea environment is that it is heavily influenced by processes on and in the adjacent land and open ocean environment. This interaction can be addressed by coupling different models as discussed below and this means that the UK shelf region cannot be modelled in isolation. A notable feature of the models described is that many have been developed through cross European partnerships.

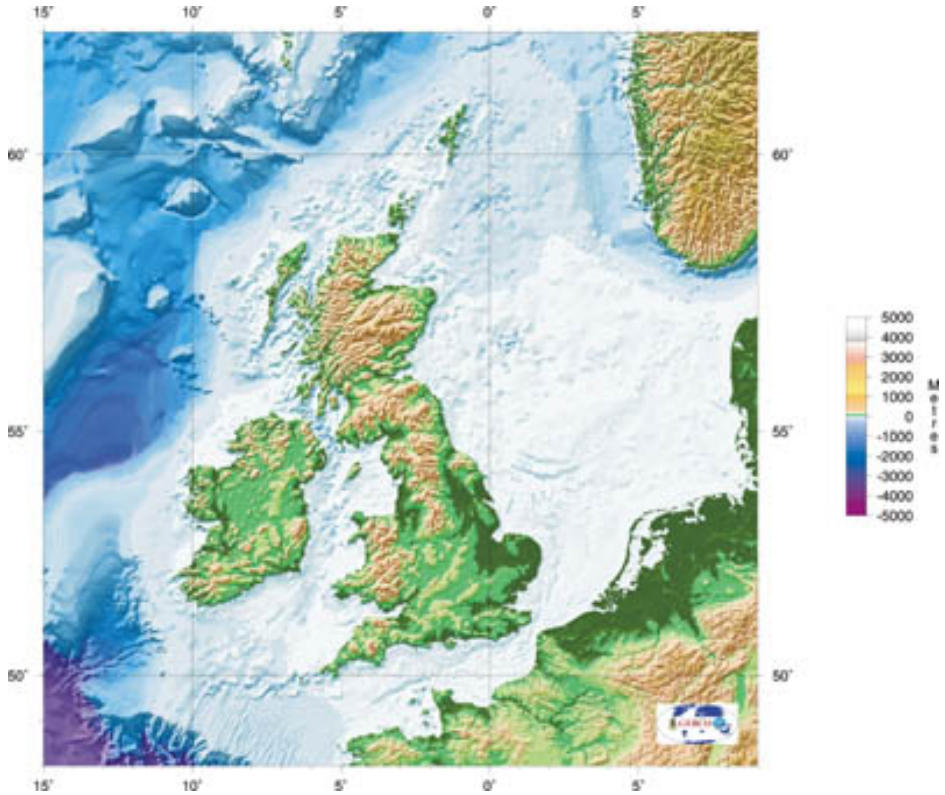


Figure 4.34. UK North East European Continental Shelf. Pale and white areas represent the coastal shelf. Source: available from NOC website: http://www.noc.soton.ac.uk/gg/images/gebco_maps/UK_SHELF_400.JPG

4.5.2 Examples of Models

We will begin with a short overview of the types of models that have been developed for various purposes in the shelf sea environment, noting their objectives and limitations, examples of their applications and anticipated future development. We then consider the extent to which these can describe the goods and services identified earlier as relevant to the UK NEA. In this overview we will in a broad sense move from more mathematically based physical models, through biogeochemical and ecological models, to less mathematically based decision support models that aim to synthesise a wide range of processes, pressures and attributes together to aid marine management. **Box 4.3.** suggests some alternative and complementary approaches for the future to the development of models for policy based applications.

Box 4.3. Lessons from Spicosa.

SPICOSA ('Science and Policy Integration for COastal System Assessment') was a European FP6 project (2007-2011) that aimed to test a method, the SAF ('Systems Approach Framework'), for advising coastal zone stakeholders and policymakers who were facing 'wicked problems' (Jentoft & Chuengpagdee, 2009), i.e. those requiring both ecological and socio-political trade-offs. For details, see: www.coastal-saf.eu, Hopkins *et al.* (2011) and Tett *et al.* (2011b). The method, which is quite different from the prevailing 'science-push' or 'policy-pull' drivers of model creation, includes two steps relevant to modelling ecosystem services and human impacts thereon.

1. Identify the issue and, in consultation with stakeholders and policy-makers, design a conceptual model of socio-ecological system processes relevant to the issue.

An *issue* was defined by the SPICOSA project as a problem involving a cause-&-effect chain from a human activity to its impact on ecosystem goods and services, in essence a social-ecological dysfunction impacting on sustainability. Examples (Hopkins *et al.,* 2011) included eutrophication (and its effects on water transparency and tourism) and shellfish population dynamics (and harvesting conflicts). The first step was not to frame the problem within a particular categorization of services and use, but to describe it through discussion with stakeholders and policymakers and agree remedial scenarios for investigation. The next step was to construct a conceptual model of the relevant parts of the social-ecological system. Many SAF applications understood this activity as a specialized technical task for modellers, but a key lesson learnt (McFadden *et al.,* 2012) was that such a model should be jointly constructed to take account of the worldviews of each group of participants. Where these worldviews conflicted, exploration of each participant's claims might either reconcile, for example, differences in scientific and stakeholder understanding, or elucidate important processes unfamiliar to one group or another. The resulting model was likely to be more reliable and more trusted by service users.

2. Use general-purpose modelling software and already-available model blocks to quickly build, test and document a simulation model based on the agreed conceptual model. Use the model to simulate scenarios agreed with stakeholders and policy makers.

This was the technical step, requiring familiarity with algorithms, programming, the getting of data for boundary conditions and parameter values, and the quality assurance of the model and its default simulations. SPICOSA examined coastal zone problems on the scale of a lagoon or fjord, cases in which it was unlikely that there would be sufficient funds for a fully-fledged research programme or the time to mount one. Thus the principle was to use existing science and data to assemble a model rapidly. The project used the commercially available ExtendSim software package (www.extendsim.com), which allows inexperienced modellers to construct reliable models by linking blocks identified by icons, providing a model that documents itself graphically. A linked aim was to develop a library of ExtendSim model blocks for key processes involved in coastal zone issues, such as algal growth (part of eutrophication), shellfish metabolism, small business economics, and so on. But although some users and stakeholders very much liked ExtendSim, aims to build appropriate models were constrained by its limited ability to handle spatial grids, and the construction of a library was not very successful. Like other packages, such as STELLA (www.iseesystems.com), ExtendSim is useful for teaching, and for stakeholder engagement in models of intermediate complexity, but not for spatially or functionally complex hydrodynamic or ecological models.

4.5.2.1 Physical Ocean Models

The first group of models are those used to forecast the transport of water, heat, salt and energy on a variety of time scales. Many of these are now operational models used routinely to provide forecasts and scenarios for the public and for users of the marine environment.

The first group of models considered here relate to sea-level itself and these include predictions of tidal height which have been routinely made with considerable confidence for many years. Certain meteorological conditions can lead to storm surges that lead to unusually high tides. There are operational models to predict such storm surges (**Figure 4.35.**) forecasts which have been demonstrated to have good accuracy (http://www.ncof.co.uk/index.htm). These models incorporate tidal predictions, detailed sea floor bathymetry and weather forecasting from the Met Office to produce forecasts of actual tidal heights and warn of storm surges (Flowerdew *et al.*, 2010). The resulting storm surge projections are provided to the UK Environment Agency to support flood management activity. Storm surge models are designed to deliver forecasts up to two days ahead, but information from this model can also be used to help predict future storm surge levels in support of the design of flood defences which have a proposed lifetime of decades e.g. Lewis *et al.* (2011).

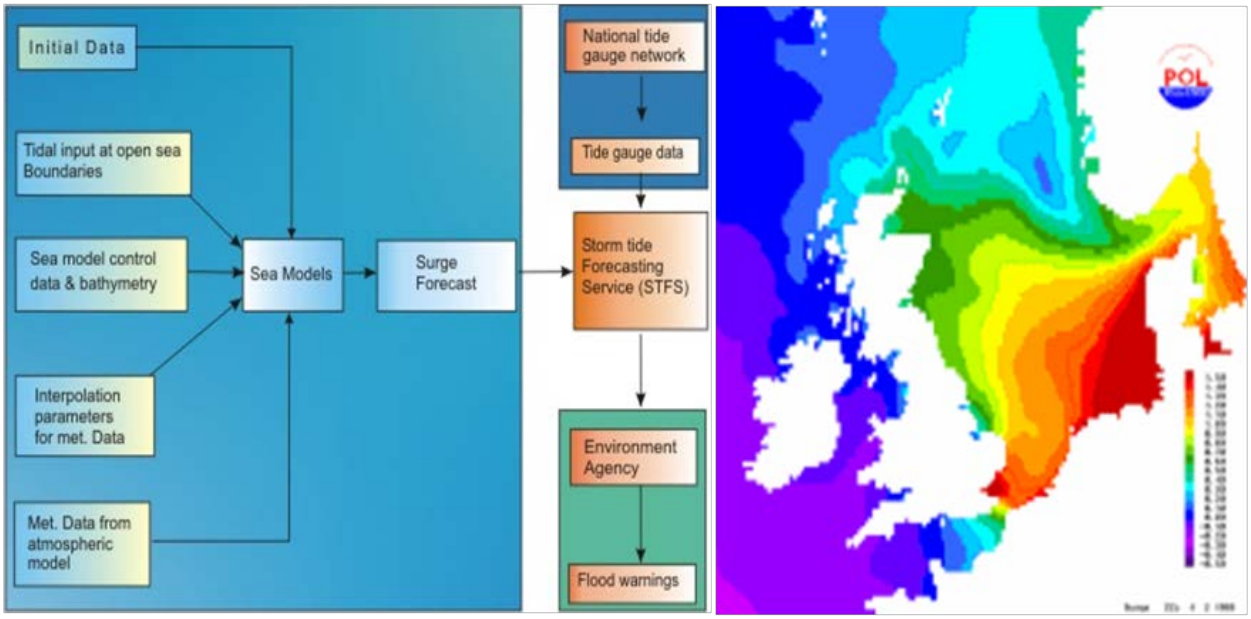


Figure 4.35. UK storm surge forecasts. Source: http://www.ntslf.org/numerical-modelling/storm-surge-model

A related issue is the general rise in sea level in the future, due to global warming, which threatens coastal areas. The model projection of future global sea level rise depends on knowledge of the changes in global sea level rise itself, driven primarily by the warming and consequent expansion of the oceans, and the inputs of water from melting glacial ice. The effects of ocean warming on sea level can be predicted quite well, but the effects of ice sheet melt over coming decades are currently rather uncertain. Future local relative sea-level rise at any particular location also depends on the changes in the land surface itself, which in the UK is primarily a response to the removal of glacial mass following the last glaciation. This deglaciation effect now leaves the north western UK rising and the south east falling at rates that are significant in terms of overall sea level. This land movement in the UK can be

estimated from extrapolations of rates measured over recent geological past or from models of isostatic readjustment. The models for prediction of UK sea level have been reviewed by Shennan and Woodworth (1992). The UK government now publishes estimates of future sea-level rise[27] and the main uncertainties in these projections (**Figure 4.36.**) relate to scientific uncertainties over the impacts of climate change on ice sheet melt and uncertainties related to future emission scenarios and their impacts on climate change (Hanna *et al.*, 2013). These projections and their associated uncertainties can still be used to estimate future flood risk by, for example, incorporation into probabilistic models (Purvis *et al.*, 2008).

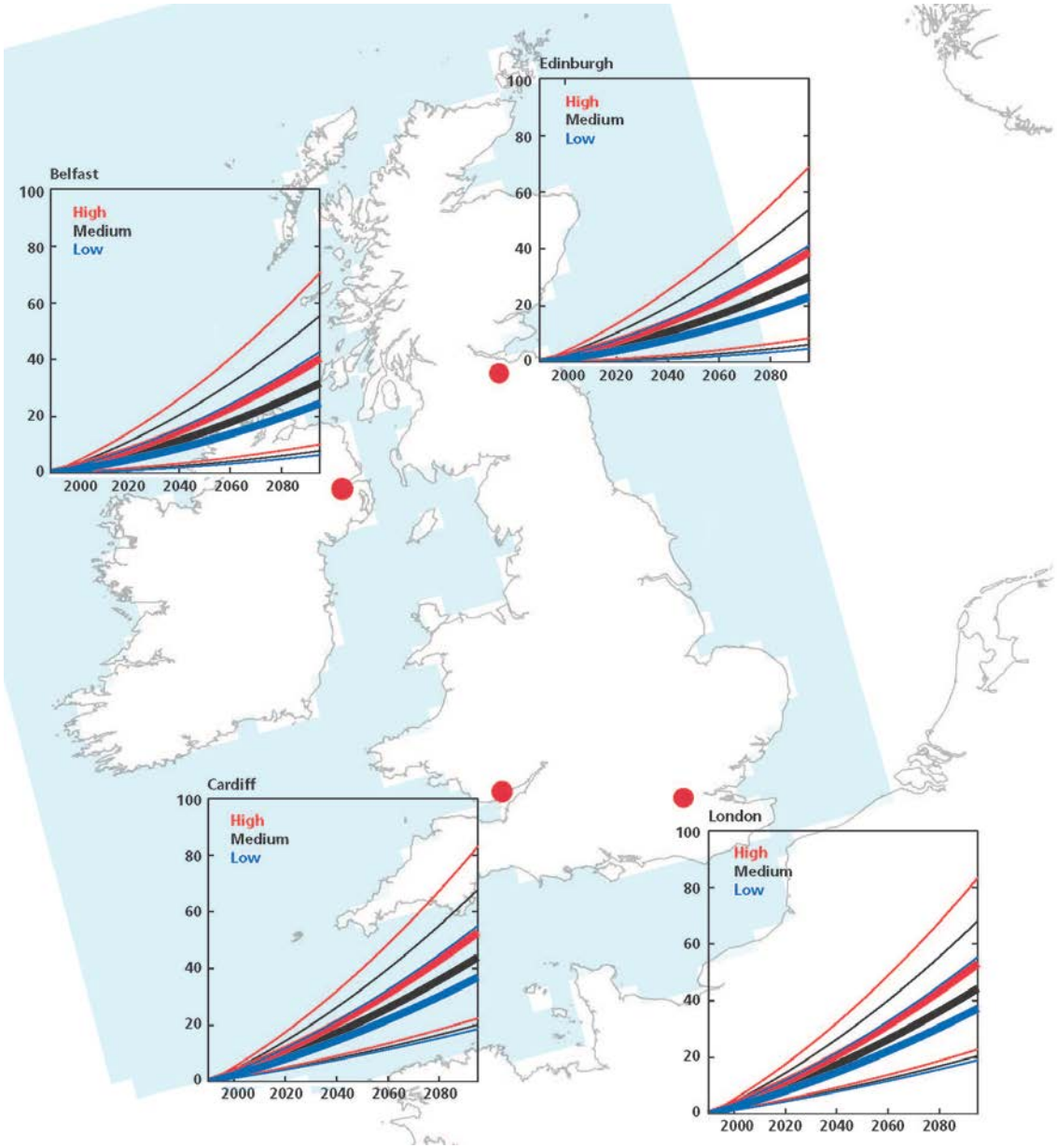


Figure 4.36. Sea level rise. Source: UKCIP: http://www.ukcip.org.uk/wordpress/wp-content/PDFs/UKCIP_sea-level.pdf

[27] http://www.ukcip.org.uk/wordpress/wp-content/PDFs/UKCIP_sea-level.pdf

A second group of models are the shelf sea hydrodynamic model systems which can be used to predict the circulation of heat, salt and water. Applications include flows of these components within the North-West European shelf sea waters, and the exchange of these properties with offshore waters of the North Atlantic. Models can also provide predictions of wave height (Cavaleri *et al.*, 2007; Chapter 5 in Lowe *et al.*, 2009), which is important particularly for offshore operations and which, when coupled to hydrodynamic models, can contribute to studies of sediment transport. Both must be driven by meteorological models and hence can be coupled to climate change scenarios. Climate change will not only induce sea level rise, but also warming of the coastal seas and hydrodynamic models offer a means to investigate the impacts of such climate change.

The main UK shelf sea hydrodynamic model system has been based on the POLCOMS system for many years (Holt *et al.*, 2009). For this application POLCOMS is linked to, and in part forced by, a wider ocean scale application of the NEMO model (see below) and either measured, long term average or modelled wind and river run off data. The quality of the output of such a model depends at least in part on the quality and accuracy of these input terms. This shelf hydrodynamic model can be coupled to other models such as the biogeochemical ERSEM model (European regional seas ecosystem model) which is described below (Holt *et al.*, 2009). The UK shelf hydrodynamic model system is now converting to the NEMO system which offers some improvements over POLCOMS (O'dea *et al.*, 2012) with a common modelling system for ocean and coastal seas. The output from these models is used to make public forecast of the impacts of climate change[28], for example of future changes in the temperature of the European Shelf and in the resultant hydrodynamics of the European Shelf Seas (Holt *et al.*, 2010) (**Figure 4.37.**). This warming of the North Sea and North Atlantic has already been linked with changes in species distribution (Beaugrand *et al.*, 2013). Hydrodynamic models can also support marine habitat studies providing information for instance on bioclimatic zones and the connections between regions via currents which is relevant to processes such as larval dispersion.

These hydrodynamic models can also be used to predict changes in water column seasonal stratification with rising temperatures (**Figure 4.37.**), a process that is important because stratification is coupled to the development of lower oxygen conditions in near-bed waters (e.g. Queste *et al.*, 2012). Lowe *et al.* (2009) conclude that both the intensity and duration of such seasonal stratification is likely to increase in the future in some parts of the shelf seas, but the uncertainties in the projections in **Figure 4.37.** arise from the assumed future GHG emission scenarios, model uncertainties and natural variability (Hawkins & Sutton, 2009). Hydrodynamic modelling of shelf seas is challenging for many reasons, but the complex nature of the bathymetry of shelf /ocean boundary and the associated complexity of the shelf/ocean water exchange is a particular challenge (Huthnance, 1995). NERC have recently begun a substantial research programme[29] aimed particular at a better understanding of this complex boundary.

Hydrodynamic models can also be used to estimate the transport and deposition of suspended sediments within the shelf seas due to natural processes of tide and wind, and also by activities such as dredging and fishing (e.g. Luyten *et al.*, 1999; Lee *et al.*, 2002; van der Molen *et al.*, 2009). These models require predictions of hydrodynamics to be coupled to descriptions of different sediment types, since the movement of coarse sand and cohesive muds by ocean currents differ in important ways. Sediment resuspension affects the light climate and hence primary production, the transport and fate of sediment bound pollutants, sediment carbon burial, and the nature of the seabed itself, which is a critically

[28] http://ukclimateprojections.defra.gov.uk/
[29] www.sams.ac.uk/fastnet

important for benthic ecology. The incorporation of sediment resuspension and deposition processes within hydrodynamic models also allows the water column transport and the development of bedforms to be predicted (e.g. van der Molen *et al.*, 2004; 2009; Dolphin & Vincent, 2009). Models are also available to describe processes of beach erosion and deposition (e.g. Bacon *et al.,* 2007) in support of management of coastal sea defences.

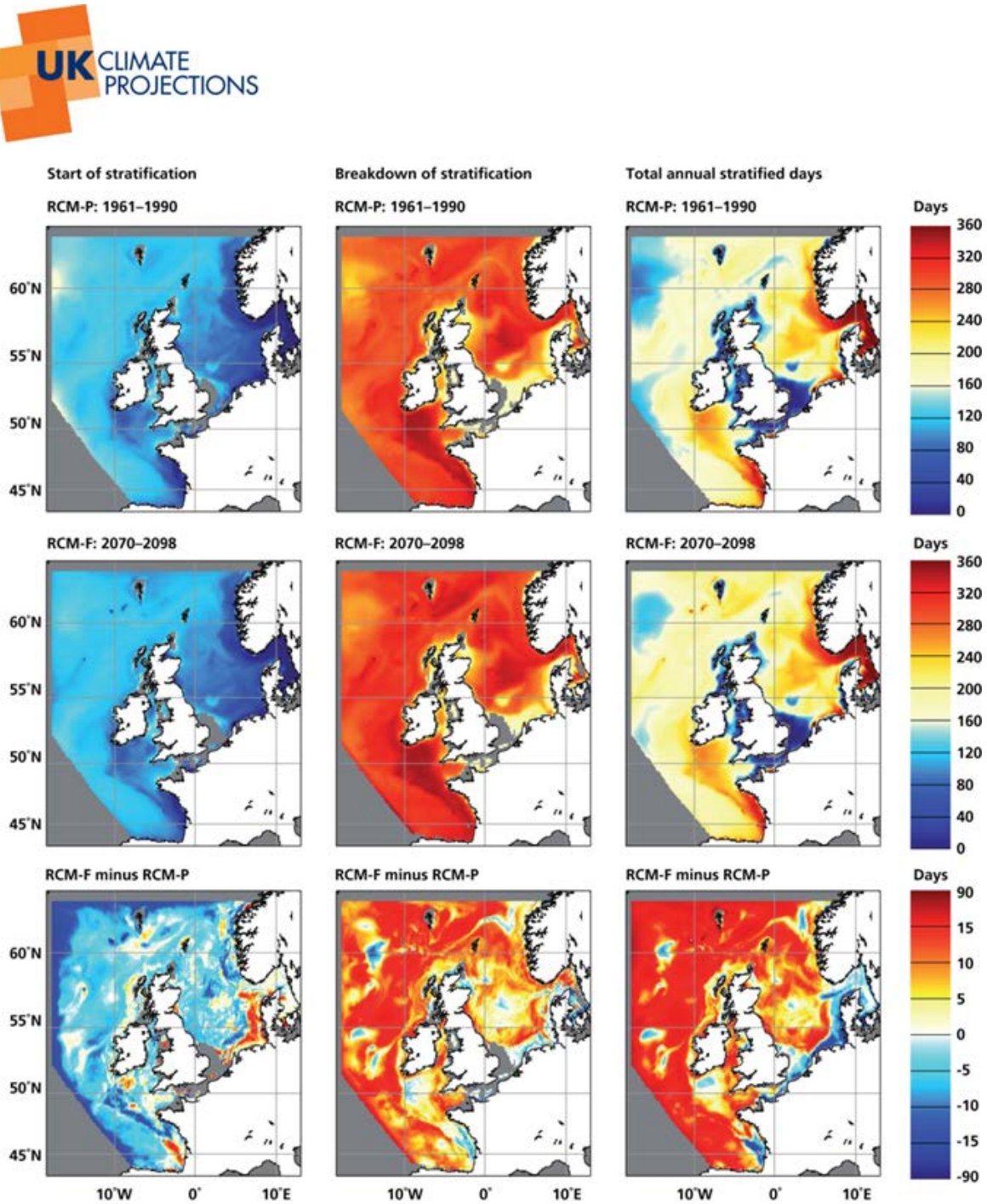


Figure 4.37. Changes in North Sea stratification. Source: Lowe *et al*. 2009. Figure 6.10 http://ukclimateprojections.defra.gov.uk/23022.

4.5.2.2 Biogeochemical and Ecological Models

Models of the lower trophic levels of the marine food web require coupling of light and nutrient supply, sediment water interactions and the ecology of phytoplankton, bacteria and zooplankton. These are mechanistic models (i.e. based on process understanding) driven by meteorological forcing, open boundary forcing (representing far-field influences) and nutrient forcing (from land, ocean and atmospheric sources). These models are governed by the hydrodynamic conditions, and therefore only represent the (lower) trophic levels, for which the movement of the relevant organisms is dictated by the currents (such as plankton). Relatively simple biogeochemical models are available which are suitable for addressing particular issues (e.g. CSTT model, **Box 4.A1.**) as well as large more complex modelling systems. One of the best developed and most extensively used of such large complex biogeochemical modelling systems in the UK and Europe is ERSEM (**Figure 4.38.**, Baretta *et al.*, 1995; Baretta, 1997; Blackford *et al.*, 2004; Edwards *et al.*, 2012). This model describes the rates of a wide variety of processes, but the model is limited to a small number of general classes of ecological groups; for example four different broad classes of phytoplankton, based on their ecological function. The models can be run in isolation or within a hydrodynamic model (e.g. van Leeuwen *et al.*, 2012). These kinds of models can be used to consider the effects of future climate change on the lower marine trophic food web and also the impact of changes in nutrient inputs (e.g. Lenhart *et al.*, 2010; Holt *et al.*, 2012; Artioli *et al.*, 2013). In an inter-comparison exercise for the North Sea, different models produced similar but not identical results, reflecting the differences in model design, and emphasising that such models are valuable, but currently limited, tools for describing the marine ecosystem and developing marine management policy (**Figure 4.39.**). The degree of uncertainty in such models can be rigorously assessed (de Mora *et al.*, 2013) Models have been shown to have skill in simulating several ecosystem components particularly at the coarser scale (Artioli *et al.*, 2012; Shutler *et al.*, 2013). A large scale NERC/Defra programme on Shelf Sea Biogeochemistry[30] is just beginning and this offers the prospect of further improvements in our understanding and modelling of the UK shelf seas. This programme will focus on developing the EREM/NEMO system.

The multi-model-ensemble approach, resulting from international collaboration, is a key part of the IPCC strategy for assessing uncertainty in predictions of climate change (Meehl *et al.*, 2007) and this approach could usefully be widely adopted where possible for predicting change in marine ecosystem services where these are part of a shared marine region or subregion (such as the Greater North Sea under the MSFD), such as in Lenhart *et al.* (2010).

The inputs of nutrients to the coastal waters come from offshore, rivers and the atmosphere, and the management of such inputs is an important component of marine ecosystem and socio-economic management. Groundwater inputs may be locally important but are not specifically considered here. The offshore supply is currently estimated from modelled flows of water between the ocean and shelf (see above) coupled to nutrient climatologies based on observed (rather than modelled) distributions of nutrients. River inputs data for ERSEM model runs and for the reporting of UK inputs to international bodies such as OSPAR is based on gauged river inputs. These chemical inputs are sampled at rather low frequency compared to their known short-term variability and in some regions of the UK sampling stations are a considerable distance inland of estuaries to avoid problems of operating gauging stations in areas of tidally reversing flow (Littlewood & Marsh, 2005). This gauging station issue requires

[30] http://www.nerc.ac.uk/research/programmes/shelfsea/

adjustments to account for input and removal processes taking place below the final gauging station to provide accurate representations of inputs to the estuary itself.

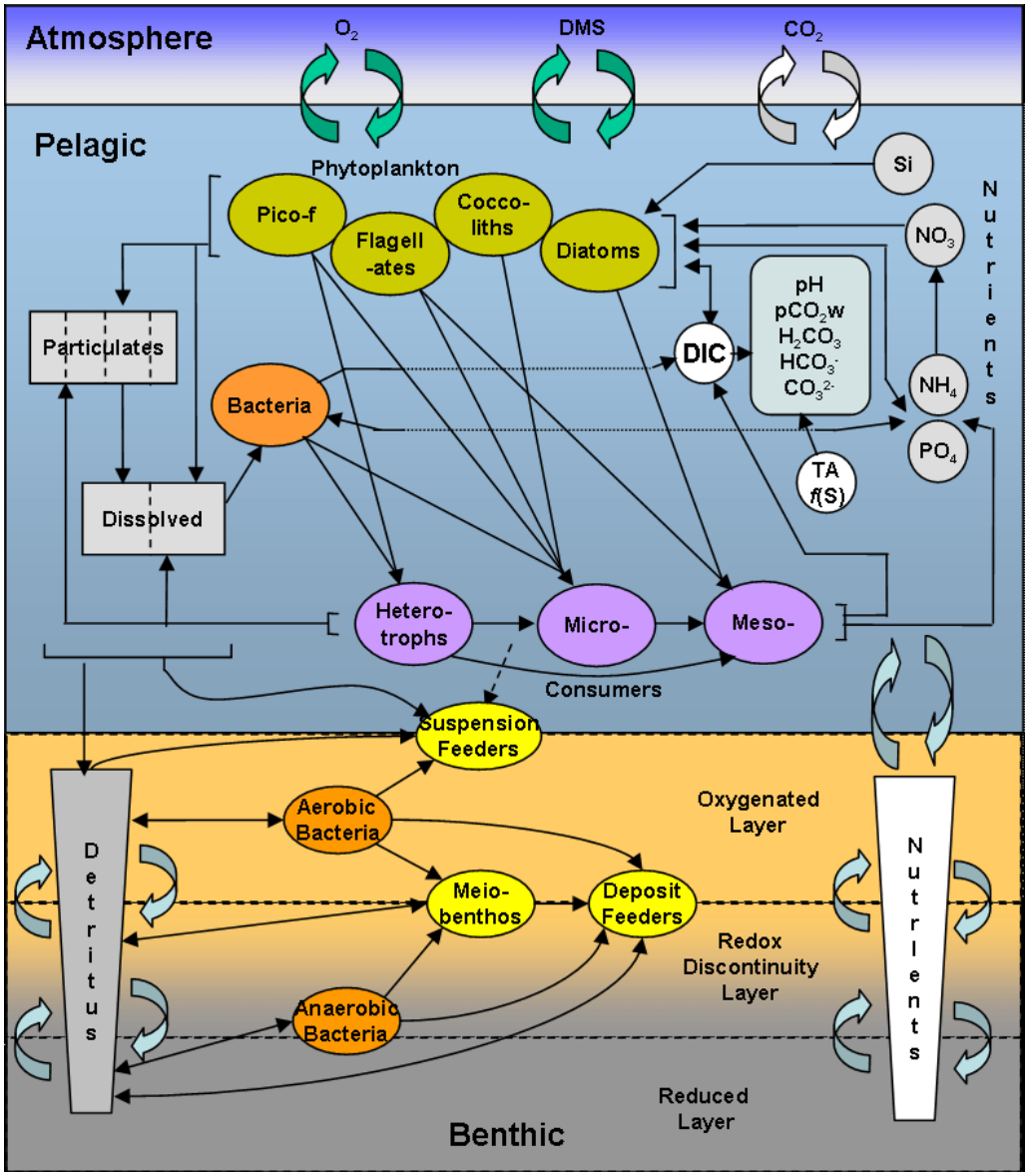


Figure 4.38. ERSEM. Source: MEECE website. http://www.meece.eu/library/ersem.html

An alternative approach to using the monitored river inputs is to model them. Sophisticated models of catchment nutrient flows are available (e.g. SWAT (Gassman *et al.*, 2007)) and these types of models will be developed further within the UK as part of the Defra Test Catchments[31] and NERC/Defra macronutrient cycles[32] programmes. However, the detailed nature of these models (making them very demanding of data and computer resources), makes it difficult to fully couple these to shelf sea models, but this is now becoming possible in systems where a few large rivers dominate (Lancelot *et al.*, 2007). There are simpler models available to estimate global scale river nutrient fluxes, based on simplified

[31] http://www.lwec.org.uk/activities/demonstration-test-catchments
[32] http://www.nerc.ac.uk/research/programmes/macronutrient/

assumptions about inputs and generalisations about nutrient processing in catchments (e.g. Seitzinger *et al.*, 2010). However, where directly gauged flows are available, such as in the UK, these are probably preferable to model derived flows.

Despite the uncertainties over inputs, models offer a method to evaluate the impacts of inputs that cannot be done in any other way. **Figure 4.40.** illustrates the impact of particular groups of rivers on North Sea nutrient levels, using a model "experiment" in which particular rivers are "tracked" in the model so that their nutrients can be identified within the marine ecosystem, throughout the chemical and biological cycles, and the spatial effects of the rivers on the ecosystem evaluated (Lenhart *et al.*, 2010).

The inputs of riverine nutrients to the coastal seas are modified by estuarine processes (Statham, 2012). While there is abundant evidence of modification of fluxes within estuaries, the scale and nature of these effects are poorly understood and generic models for these interactions are not available. Corrections for estuarine processes are therefore usually either based on specific models designed for a particular estuary, or on average transmission factors for each nutrient in estuaries in general.

Atmospheric inputs of nutrient nitrogen (but not P and Si) to coastal seas can be significant (of the order of 25% of land based inputs for example) and can be estimated either from extrapolation of coastal data or from models of atmospheric transport and deposition such as EMEP (http://www.emep.int/), or very rarely from direct measurements over the coastal waters (e.g. Spokes & Jickells, 2005).

The distribution and cycling of contaminants in coastal seas can be modelled, provided input data are available and the reactivity of the contaminants can be described for the hydrodynamic model. Tappin *et al.* (2008) successfully described the distribution of various trace metals in the North Sea based on published inputs and a distribution coefficient to describe the partitioning of the metals between suspended sediments (which were modelled) and the water phase.

Results from hydro-biogeochemical models can be linked to higher trophic level models (representing animals which control their own movement, such as fish) to assess the possible impact of bottom-up pressures like climate and nutrient availability, on, for instance, fish biomass or fisheries yield. Higher trophic level models can be size-structured (based on size characteristics, e.g. Blanchard *et al.,* 2009) or food web models (based on species characteristics). These models incorporate top-down pressures like fishing effort and fisheries management. In combination with a hydro-biogeochemical model these models can be used to assess the relative impact of bottom-up and top-down pressures on economic activities and management strategies .These models seem to be able to explain changes in fisheries community structure as a result of fishing pressure providing some confidence in their predictions of the impacts of possible future fisheries management practices. Specific models are routinely used in fisheries management and are able to consider individual species and the impact of fisheries practice on the fish stock (e.g. Mackinson *et al.,* 2009; Heymans *et al.*, 2011; and the Ecopath and Ecoprism models, see www.ecopath.org/).

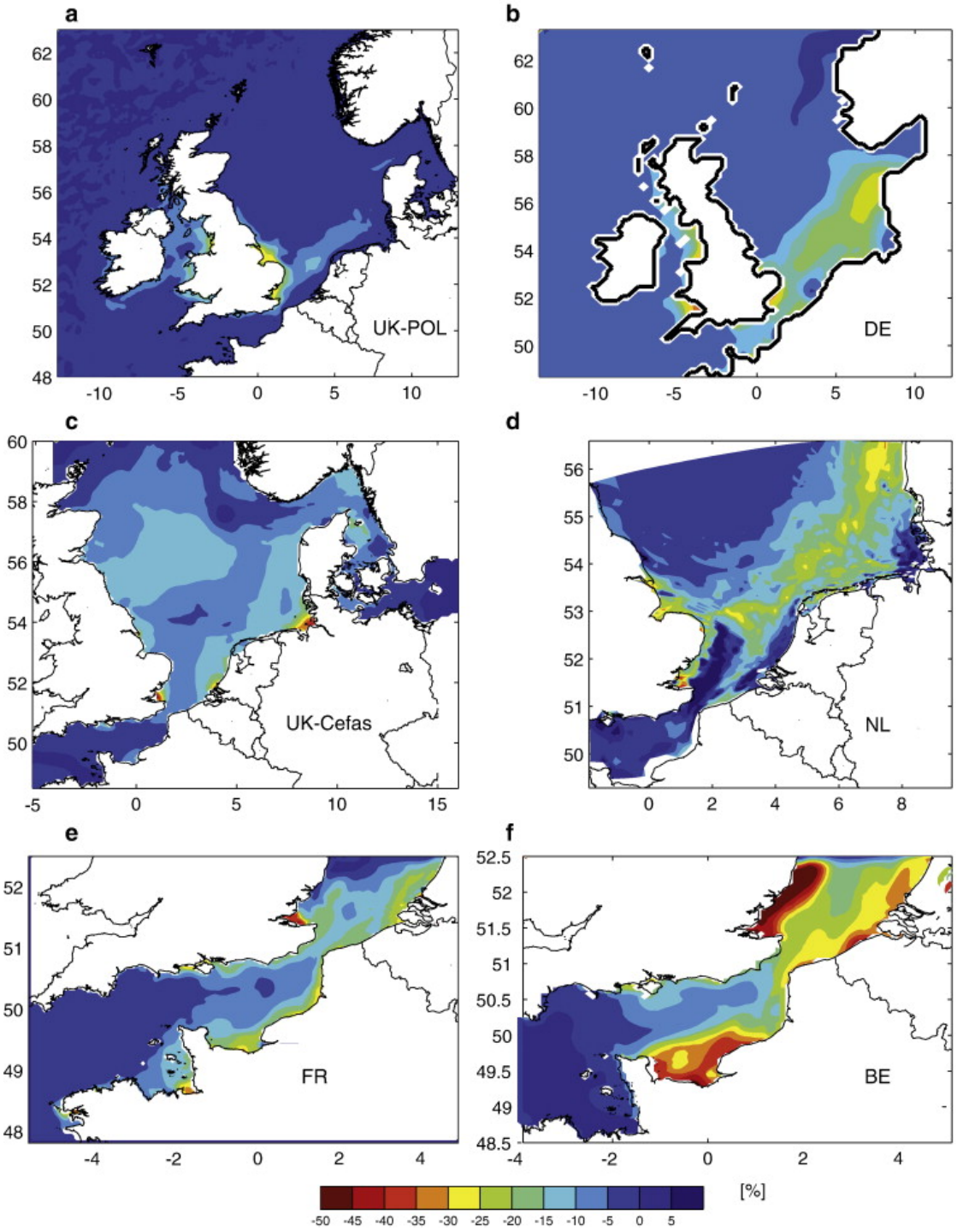


Figure 4.39. Horizontal distribution of the percentage difference in net primary production. Obtained from the 50% reduction run compared to the standard run from the six different European models including two based on ERSEM (a and c): a) UK-POL; b) DE, c) UK-CEFAS, d) NL, e) FR, and f) BE model simulations. Source: Lenhart *et al.* (2010). Reproduced with permission of Elsevier.

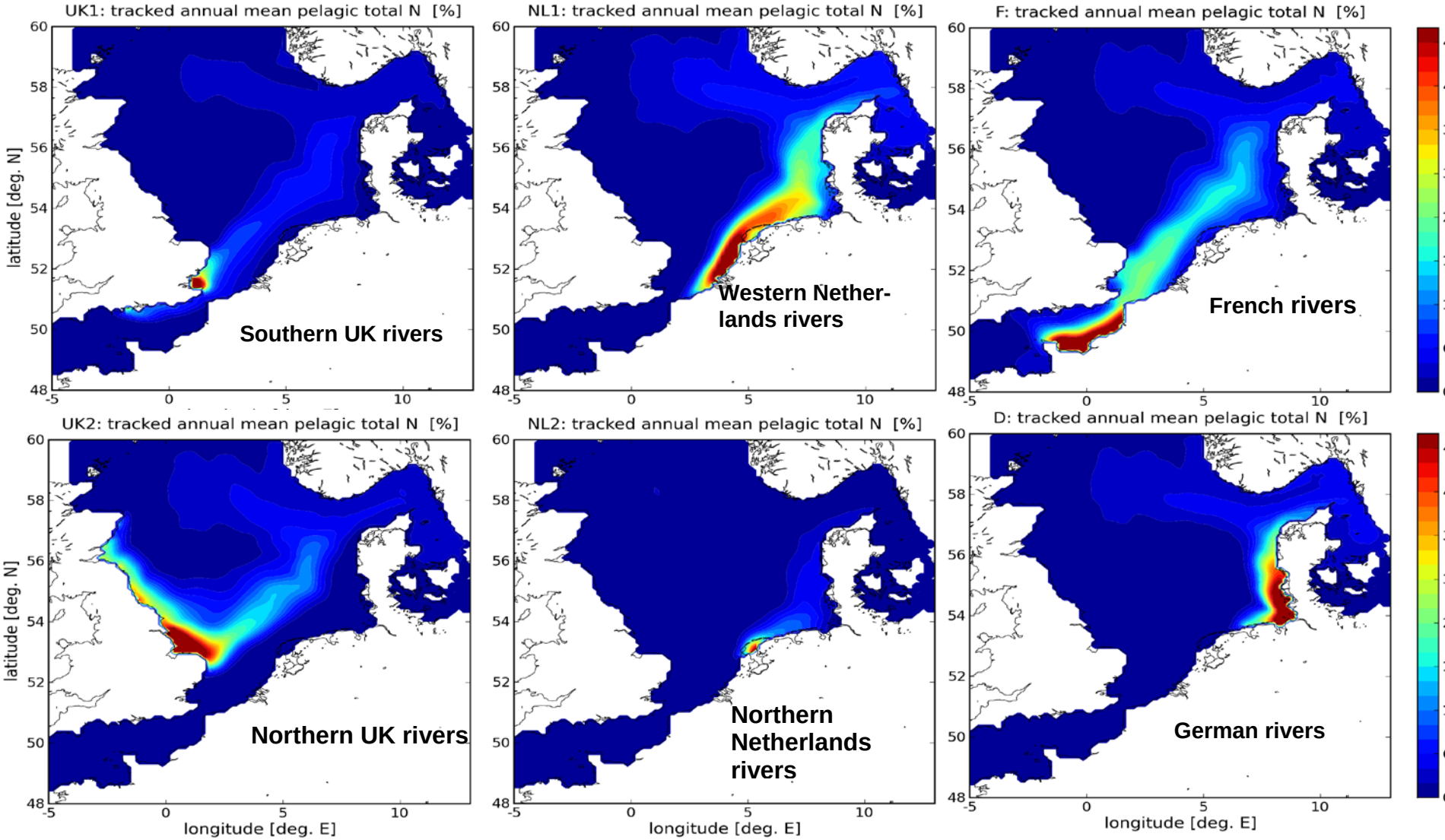


Figure 4.40. Contribution of different river systems to total fluvial N load to the North Sea.

Temperature changes will also affect the distributions of fish in coastal waters, as the fish optimise their temperature and habitat preferences which allows for habitat suitability and climate change impact models to be developed. Jones *et al.* (2012) have reviewed the outputs of three such models and concluded that the models are useful, but need to be used with reference to the model uncertainty. They therefore require additional expert judgement to deliver effective management advice. The overall effects of climate change and other environmental pressures such as ocean acidification and hypoxia on the whole of the fish community and the commercial fishing industry have been evaluated using models. The outputs of different models give rather different results reflecting assumptions about the interactions of species, but Cheung *et al.* (2012) used one of these model systems to suggest significant changes in fish landings by 2050 with considerable financial implications (**Figure 4.41.**). These models have been also expanded to consider interactions between species based on primary production spatial and temporal availability (Fernandes *et al.,* 2013a).

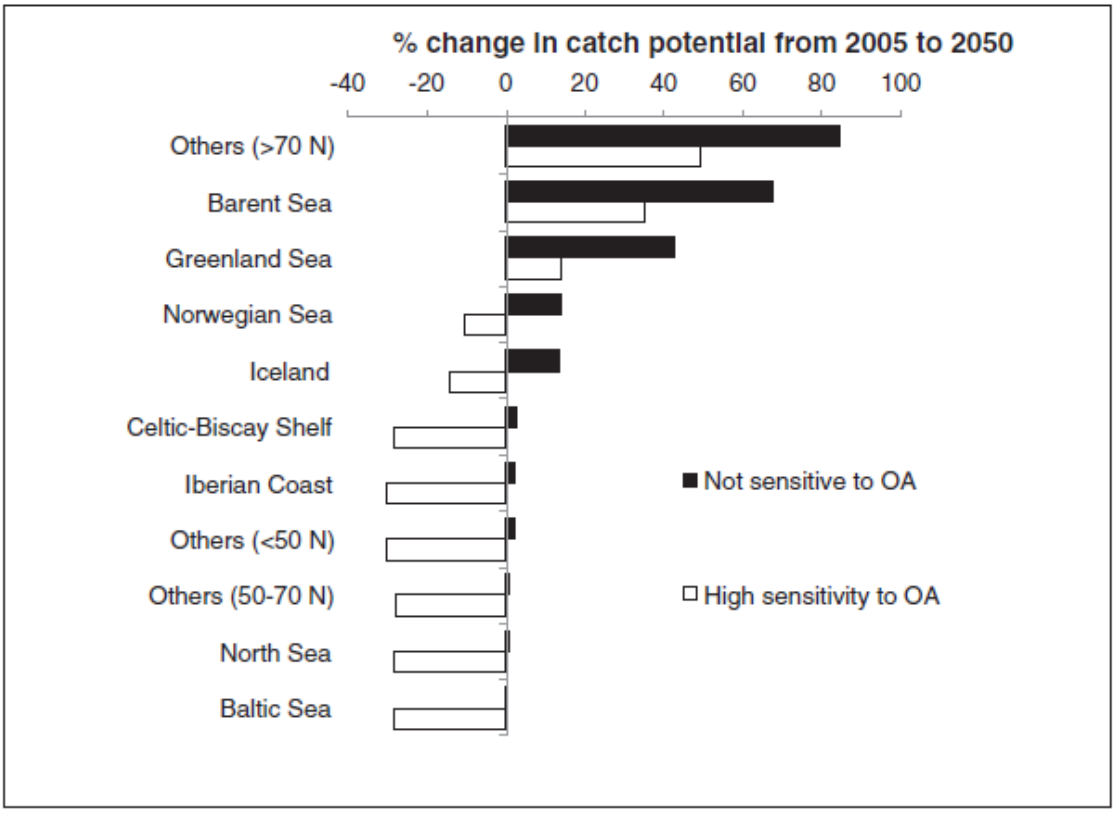


Figure 4.41. Projected changes in fish catch for different European regions from 2005 to 2050 due to climate change alone (black bars) and with the additional pressure of climate change (white bars). Source: Cheung *et al.* (2012). Copyright © 2012 John Wiley & Sons, Ltd.

End-to-end models offer the opportunity to incorporate all relevant ecosystem and physical processes into the modelling system, including anthropogenic pressures. ERSEM is potentially able to work as a component of an end-to-end model, although it is most widely used for the lower trophic levels. The scale and complexity of the processes involved create major challenges for the construction and validation of the models and hence simplification and parameterisations are required. The key is to ensure that such simplifications are appropriate to the goals of the work. ATLANTIS is an example of an end-to-end model that is adaptable to tackling different tasks and can provide valuable information about interactions across the whole ecosystem from nutrient cycling to fish (Fulton, 2010; Link *et al.*,

2010). Heath (2012) has recently applied such an end to end modelling approach to North Sea fisheries yields. Although this model cannot consider individual species, it does reveal the complex interplay between groups of fish (e.g. pelagic and demersal) and other components of the environment, and provides important information to support environmental management. The complexity of the whole marine ecosystem is such that there are currently no species level marine ecosystem models, although there are models for particular species (Hjøllo *et al.*, 2012). The new NERC Marine Ecosystems Research Programme aims to improve this situation.

The biogeochemical models described so far are all designed to operate basically at the scale of the whole shelf sea. However, for the operation and regulation of commercial activities such as fish farms, two sorts of smaller-scale models are needed. At the fish farm scale itself, models with a high spatial resolution, of the order of tens of meters or better, such as the particulate waste distribution simulator DEPOMOD (Cromey *et al.*, 2002), are required by commercial operators and regulators. On the water-body scale, simple models such as that for 'Equilibrium Concentration Enhancement' (Gillibrand & Turrell, 1997) and models of intermediate complexity, such as ACExR-LESV (Tett *et al.*, 2011a) are useful for estimating the capacity of lochs and estuaries to assimilate farm waste.

4.5.2.3 Bayesian Belief Networks and Decision Support Models

As the ecosystem representation and the associated models become more complex it becomes increasingly difficult to develop mechanistic models that quantitatively describe all the interactions of interest. The outputs from such large and complex model systems can also often be difficult to interpret. The complexity of such models and their outputs can also limit their utility for environmental management. This has led to the development of alternative modelling and synthesis approaches that are designed to work where knowledge is incomplete and where very different sorts of information, including expert judgement as well as quantitative mechanistic or correlational relationships, need to be integrated. Examples of such approaches include Bayesian Belief Networks (probabilistic graphical models) and database or spread-sheet based integrative models.

Bayesian Belief Networks (BBNs) are models that graphically and probabilistically represent causal and statistical relationships among variables (McCann *et al.*, 2006). BBN models are flexible integrative modelling tools, which can incorporate quantitative information that can be obtained from other models, empirical data, monitoring or specific investigations. Where data is missing, qualitative information (mostly from expert judgement) can be applied, so that the BBN becomes a flexible integrative modelling tool. The BBNs generated outputs reflecting uncertainty, and can also clearly document where the assumptions are made, making them a very good tool for analysis of relationships between different components and management options (Jensen & Nielsen, 2007).

BBNs can also deal with a wide range of problems to support decisions in environmental management, natural resources and ecosystem services, e.g. Varis and Kuikka (1997), Marcot *et al.* (2001), Casteletti and Soncini-Sessa (2007), Henriksen *et al.* (2007), Uusitalo (2007), Barton *et al.* (2008; 2012), Johnson *et al.* (2010), Haines-Young (2011), Chen and Pollino (2012), Fernandes *et al.* (2010, 2012, 2013b), Johnson & Mengersen (2012) and Landuyt *et al.* (2013). BBNs are also a valid tool for participatory environmental modelling with experts and stakeholders (Bromley *et al.*, 2005; Henrinksen *et al.*, 2007) and can effectively integrate environmental and socio-economic considerations (Barton *et al.,* 2012).

The word 'belief' in ‘BBN model' emphasises that models are human societal constructs. As exemplified, in an application (Langmead *et al.,* 2009) to the state of the north-western Black Sea as a function of

land-use (in the Danube catchment) and fisheries, a BBN model has two components. One of these was a conceptual DPSIR model developed in expert workshops, specifying the links between key processes for which indicators were available. This can be seen as a mechanistic, albeit qualitative, model. The second component was empirical, involving Bayesian analysis of indicator time-series that specifies probability distributions for effect variables given frequency distributions for cause variables, which were de-dimensionalised by assigning to a small number of state categories (e.g. 'low' or 'high').

Bayesian models can be based on identifying the relationships between ecosystem components as well as statistical information about those components, and then allowing the modelling software to develop the probability relationships between the components of interest. This approach has the advantage of including uncertainty estimates within the output information and lack of knowledge about particular components. However, this approach cannot model feedbacks within the system well, and the lack of mechanistic descriptions of processes means that dynamic variability in space and time within a system cannot be modelled (Langmead *et al.,* 2009; Landuyt *et al.*, 2013). BBN models are relatively new but are proving valuable in dealing with complex marine management issues, in particular because they (i) can incorporate expert judgement where detailed mechanistic relationships are poorly known, and (ii) because their output includes uncertainty estimates (e.g. Langmead *et al.*, 2009). BBNs can be developed as dynamic models where temporal variability is integrated. There are also options to integrate modelling from BBNs and Geographic information systems (GIS) allowing spatial analysis and representation of BBN models outputs in a map (Barton *et al.,* 2008; Li *et al.*, 2010; Stelzenmuller *et al.,* 2010; Johnson *et al.*, 2012). Franzen *et al.* (2011) used the SAF method (**Box 4.3.**) to construct a model for eutrophication in the Himmer fjord, near Stockholm. The model, of intermediate complexity, combined simple mechanistic models for estuarine exchange and nitrogen cycling, with a regression model for the relationship between the concentration of total nitrogen in the fjord's water and the Secchi depth during summer. The Secchi depth, a simple measure of water clarity, increases with water transparency, and the social benefits were estimated from a study of WTP for transparency as a sign of good quality water. This approach illustrates how ecosystem services benefits might be brought into models.

Another alternative approach to synthesising multiple forms of complex information (such as a mixture of quantitative, qualitative or based on expert judgement) is via spread-sheet or data base tools and this can also be set within a spatial context using GIS. Such a system might allow, for example, the pressures and features on a particular area of seabed to be drawn together to allow a management to identify if a management response is necessary and if so what that should be (see **Figure 4.42.**). Such an approach has the advantage of incorporating a wide variety of information of very different types into a spatially explicit format in a way that can be directly interrogated by a manager who does not have direct experience of the model development (Net Gain, 2011). The system does not necessarily incorporate uncertainties (unlike BBNs) nor is it dynamic or mechanistic and it cannot explicitly include feedbacks (such as ERSEM), but can handle large and complex amounts of information within a geographic framework, but BBNs and mechanistic models can be combined (Andonegi *et al.*, 2011).

Both BBNs and the spread-sheet approaches offer a very valuable way to integrate and present complex ecosystem information to support environmental management and to complement and support expert judgement, particularly where knowledge is incomplete as is almost inevitably the case when trying to look across the whole ecosystem. These approaches complement rather than replace the more mechanistic models.

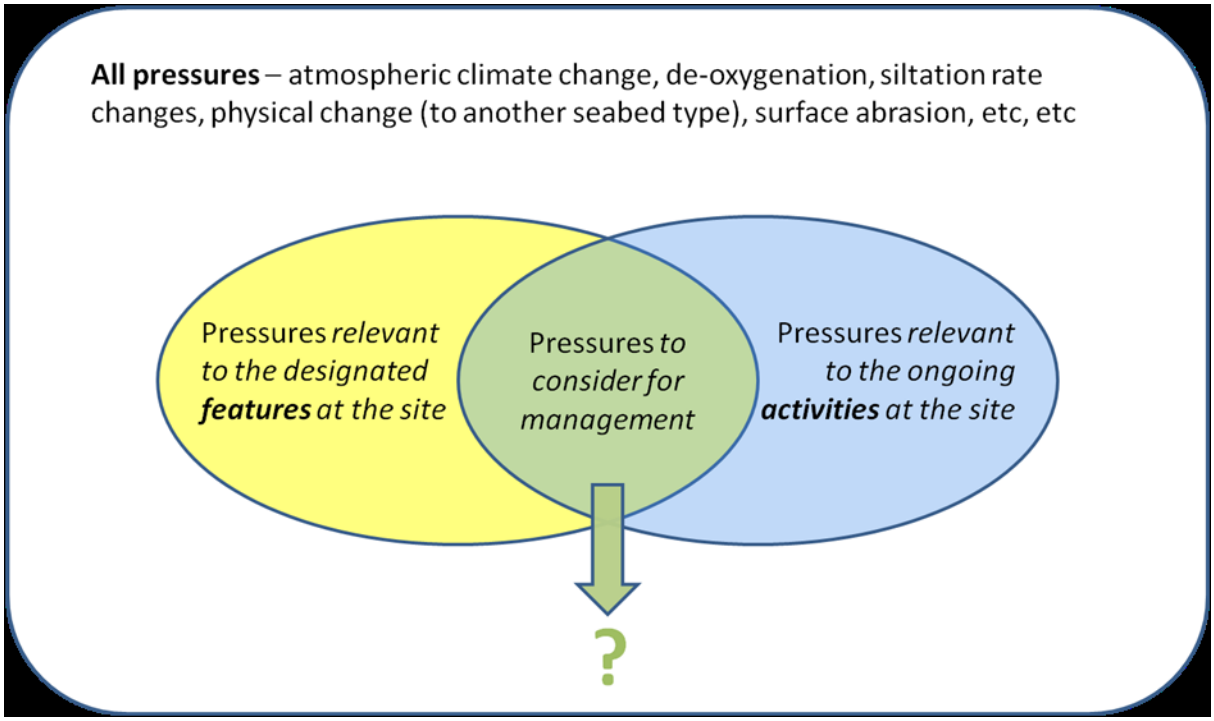


Figure 4.42. Illustrative example of database or spreadsheet modelling data sets that might be merged to provide management information.

4.5.3 Using models to describe ecosystem services and the goods and benefits derived from them.

In **Table 4.8.** we tabulate selected final ecosystem services and the goods and benefits and link them to the types of models that can contribute to the effective management of these services. The table is not designed to be an exhaustive listing of models, but rather is designed to illustrate that a wide range of models are required spanning a variety of scales and complexities to achieve this wide variety of goals. As noted at the beginning of this section there is no single group of models suitable for this task, but rather a wide variety of tools are required in support of expert judgement and for some services, models are not really available.

Table 4.8. Final ecosystem services, goods and benefits and models that can help provide information on these.

Final Ecosystem Services	Goods and Benefits	Types of models
Fish and shellfish	Food	Wild Fisheries – fisheries yield models, biogeochemical models, end-to-end, hydrodynamic and sediment transport models, climate change models, integrative tools –Bayesian networks and spread-sheet. Aquaculture – biogeochemical models, farm and water-body scale models
Algae & seaweed	Fertiliser	Macro-algae models, biogeochemical models
Ornamental material	Ornaments	
Genetic Resources	Medicines and blue technology	
Climate Regulation	Healthy climate	Biogeochemical models (C sequestration), climate change models, hydrodynamic models, Bayesian networks and spread-sheet.
Natural Hazards	Prevention of coastal erosion and sea defences	Storm surge models, sea level rise models, sediment transport models, hydrodynamic models, Bayesian networks and spread-sheet.
Clean water and sediments		Biogeochemical models, hydrodynamic models, sediment transport models, land use models, Bayesian networks and spread-sheet.
Places and seascapes	Tourism, spiritual and cultural well-being, aesthetic benefits, education	Bayesian networks and spread-sheet, Models of intermediate complexity built using the approach of **Box 4.3.**

4.5.4 Links to NEA land use change model

Our survey of modeling capabilities has shown that while significant progress has been across a range of environmental contexts, the efforts so far to link terrestrial catchments to coastal and marine environments have been limited. While estuaries are important components in such a 'coupled' approach their complexity is such that the models that do exist are site specific, or are 'box' models without spatial representation. We take a closer look below at how progress might be pragmatically made to better link nutrient flows form catchments to estuaries and coastal waters and the consequences for ecosystem services provision.

Marine hydrobiogeochemical models describe the hydrodynamic conditions, nutrient cycling and lower trophic levels of the marine environment. They are mechanistic models (i.e. based on process understanding) driven by meteorological forcing, open boundary forcing (representing far-field influences) and nutrient forcing (from land and atmospheric sources). These models are governed by the hydrodynamic conditions, and therefore only represent the (lower) trophic levels for which movement is dictated by the currents (such as plankton). Results from hydrobiogeochemical models can be linked to

higher trophic level models (representing animals which control their own movement, such as fish) to assess the possible impact of bottom-up pressures like climate and nutrient availability, on for instance fish biomass or fisheries yield. Higher trophic level models can be size-structured models (based on size characteristics) or food web models (based on species characteristics). These models incorporate top-down pressures like fishing effort and fisheries management. In combination with a hydrobiogeochemical model these models can be used to assess the relative impact of bottom-up and top-down pressures on economic activities and management strategies.

At Cefas the main hydrobiogeochemical model is GETM-ERSEM (van Leeuwen *et al.*, 2013). This model, which has been set up for the North Sea, Irish Sea, Channel and the European Shelf, is driven by ERA40 meteorological data from the ECMWF and by daily riverine nutrients from the OSPAR ICG-EMO database (Lenhart *et al.*, 2010). Cefas currently applies climatologies on the open boundaries, so that far-field influences are not represented. This could be improved by using hydrodynamic and nutrient data (e.g. Holt *et al.*, 2012). Atmospheric deposition of nutrients is not included, but is deemed a minor input compared to land-based sources. The GETM-ERSEM model has been linked with the higher trophic level size-structured model by Blanchard *et al.* (2009), so that dynamics of the lower trophic levels feed through to the higher trophic levels. The fish biomass model represents both pelagic predators and benthic detritivores, and so captures the two main food paths in a simple size-based approach.

Although there are now coupled marine lower and higher trophic level models, a link between the terrestrial and marine environment is still challenging. Marine food webs are driven by the availability of light and nutrients, and in coastal areas the nutrient levels are determined by land run-off and riverine sources (off-shore nutrient levels are determine mainly by oceanic conditions, see Lenhart *et al.*, 2010). Observations of riverine nutrients describe past loads, but are not capable of predicting future loads into the marine environment based on future rainfall predictions. They also do not include direct land run-off, as they cover only 63% of the UK landmass (Littlewood *et al.*, 2005). Observational gaps, reductions in monitoring effort and a time lag of 2-3 years between observations being made and becoming available further complicate matters. A terrestrial land-use model (like the UK NEAFO model or the ehype land surface model[33]) able to predict nutrient loads going into the marine environment would be better placed to estimate current and future nutrient inputs. Use of such a model would significantly improve the confidence in marine response predictions to future climate forcing and facilitate research where specific observations and simultaneous model simulation should provide insight in natural processes. Such a coupling between the terrestrial and marine environment would also allow for inclusion of estuarine processes, which can recycle up to 50% of river borne nutrients. Estuarine processes are currently neglected as riverine observations are directly applied to the marine environment.

There are several options to link the terrestrial and marine environment (e.g. Torres and Uncles, 2011; Uncles and Torres, 2011). Until nested models to represent land-estuary-sea dynamics are available, a simpler approach using estuarine box models is suggested. A box model is a model without spatial representation, which captures the main dynamics as a function of time and forcings. As an example, an estuarine box model predicts algae growth in some estuaries. The following options for linking land to sea are possible with varying degrees of effort.

1. A simple but effective coupling can be achieved by using results from the land-use model (flow and nutrients) as direct input into the marine model, replacing the riverine observational data. This can

[33] http://www.smhi.se/en/2.575/Hydrology/european-hydrological-predictions-for-the-environment-1.12711

be achieved with minimal effort, and would allow for a better simulation of current and future marine coastal conditions.

2. A more comprehensive approach, taking into account estuarine processes, would be to couple the land-use model to an estuarine box model based on estuarine classification (Prandle, 2010a; 2010b) and simplified nutrient processes. This would require some development of estuarine box models (mainly conceptual improvements), and could be included as part of the pre-processing of nutrient data before application to the marine model. A simplified representation of nutrient cycling could be based on the results of the Joint Nutrient Study (JONUS) programme, taking into account river flow, sediment availability, percentage of intertidal area and other estuarine characteristics.
3. A fully coupled approach would be to include the estuarine box model as an extension of the marine model, allowing for both marine and land-based influences on estuarine processes. This would require model development of both the estuarine and marine models, and would take into account any marine representation of the estuary based on model resolution (i.e. a fine scale marine model will spatially cover more of an estuary than a coarse scale marine model, causing the estuarine box model to represent a smaller area).

Model outputs can be integrated in a BBN model to further analyse possible changes on pressures (such as nutrients inputs) and ecosystem services under different scenarios. This can also further integrate a socio-economic component and feed into a valuation assessment of ecosystem services and appraisal of management options. We turn to the valuation problem in the next section.

4.6 Valuation of coastal and marine ecosystem services: a literature review

4.6.1 Objective

This section aims to assess the availability of primary valuation studies providing economic value estimates for those 'final goods and benefits' generated from coastal and marine ecosystems of particular relevance for the UK. This overview reveals the main gaps in the literature with respect to primary (monetary) valuation studies addressing coastal and marine habitats and specific ecosystem services, globally and in particular for Great Britain/UK. It builds upon and expands the evidence base collected for the UK NEA 2011 (Beaumont *et al.*, 2010; Jones *et al.*, 2011). Valuation is one of the tools of the DSS and aims to support the step of economic (and social) appraisal and valuation of options in the adaptive coastal management approach.

The assessment and valuation of ecosystem stock and flow situations is not straightforward and some goods and benefits cannot be meaningfully valued in monetary terms (those related to cultural services in particular). However, it is possible to provide monetary value estimates for many goods and benefits through the use of both accounting price and economic valuation methods. The former approach can be used to derive stock accounting price 'value' and the latter can be deployed to derive marginal economic values of changes in the flow of goods and benefits over time (see **Section 4.2.8.4**). The stock accounting value can play a useful role in the political process in terms of intuitively highlighting the importance of natural capital/wealth. It is not however appropriate for analysis supporting actual decision making on trade off choices which require, among others, marginal economic values.

Data deficiencies dictate that it is not possible to undertake a meta-regression analysis. The distribution of value estimates across habitats and services is such that this would not be a meaningful exercise. Hynes *et al.* (2013) aimed to produce a meta- analysis for a valuation study set in Ireland, but also found that there were insufficient value estimates for the different habitats and services to produce significant model estimation results.

Section 4.6 is structured as follows. **Section 4.6.2** gives some background on economic valuation of coastal and marine ecosystem goods and benefits. It lists the goods and benefits and habitats that we aimed to cover in this overview. **Section 4.6.3** describes the selection process and criteria that we applied to select relevant papers. The results are presented in **Section 4.6.4**. In **Section 4.6.5**, we assess the extent to which monetary value estimates of the most important ecosystem goods and benefits and habitat types (assessed on the basis of expert judgement) are available from the literature. This aims to answer the question how future resources on valuation research for the coastal and marine environmental could best be employed to fill the high priority gaps in the literature.

4.6.2 Economic valuation of coastal and marine ecosystem goods and benefits

4.6.2.1 Coastal and marine ecosystem goods and benefits

In the conceptual framework of ecosystem service assessment for coastal and marine areas (see **Figures 4.4. and 4.5.**), Turner *et al.* (2013) identified different categories of 'ecosystem goods and benefits':[34]

- products: Food (wild, farmed), bait and fish feed, fertiliser, etc. (provisioning);
- sea defence (regulating);
- prevention of coastal erosion (regulating);
- 'healthy' climate (regulating);
- tourism and nature watching (cultural);
- spiritual and cultural wellbeing, including non-use values (cultural);
- aesthetic benefits (cultural); and
- education and research (cultural).

The first category includes goods and benefits of provisioning services. Coastal and marine ecosystems provide not only fish and shellfish for human consumption, fish feed and bait, fertiliser and biofuels, ornaments and aquaria, medicines and biotechnology, but coastal margins are also used for grazing, the collection of wild mushrooms and berries, other crops, reed, timber and seaweed (Jones *et al.*, 2011). In many valuation studies, the supporting nursery function provided by coastal and marine habitats is valued by assessing at the proportion of juveniles that reach maturity and are caught and sold on markets. In other words, these studies value the nursery function based on its contribution to the provisioning service that it supports. Therefore, no separate category is included for the nursery function.

Coastal protection can be provided in terms of the prevention of coastal erosion when the gradual loss of land is mitigated by coastal habitats, or in terms of sea defence that reduce the risk of sea flooding and inundation related to natural hazards (see also Section 11.3.2.1 in Jones *et al.*, 2011). Coastal protection values include benefits of ecosystem services provided by areas that are prevented from being lost through the protection provided by coastal margins. Prevention of coastal erosion avoids permanent loss of land, buildings and infrastructure. Sea defence values relate to a risk reduction of flood, storm or tidal surge events that would damage infrastructure, business, the natural and historic environment, and other property, and also the risk of life. This risk reduction benefit depends on the location, depth and flow rate of the potential flood event (Environmental Agency & Defra, 2011).

Cultural values range from use values related to tourism and nature watching, aesthetic values, education and research, to goods and benefits of spiritual and cultural wellbeing. Aesthetic benefits are sometimes reflected in property values when people are willing to pay an additional price in the housing market that can be attributed to the presence of nearby environmental amenities. Expenses on education and research on coastal and marine are included to represent these uses of nature.

[34] For the purpose of this chapter, the list excludes water purification services, because in the UK the direct use of clean sea water as drinking water is limited, i.e. sea water use for water supply is very limited (UKMMAS, 2010). Other benefits of changes in water quality result in ecosystem goods and services such as recreation and amenity, raw materials or biodiversity and landscape values, which are the valued items and discussed in separate sections. Appendix 5 addresses this further. Following the UK NEA ESF, we also excluded services that relate to abiotic components of the areas, and services with negative impacts related to off-shore wind farms and invasive species.

In valuation studies, the reported economic value may correspond to the benefits derived from a bundle of goods and benefits. This is especially the case for studies that aim at capturing values of tourism, nature watching and aesthetic benefits of meaningful seascapes. The value that people attach to certain species and natural habitats and seascapes values may reflect both spiritual and cultural wellbeing and aesthetic values, and thus may contain an aspect of non-use (bequest, existence) values. In such cases, it is difficult to assign separate values to each of the ecosystem goods and benefits.

4.6.2.2 UK coastal and marine habitat types

In the UK NEA 2011, six coastal margin and six marine habitats were identified. The coastal margins categories are habitat based, and the marine habitats are based on sedimentation. For the purpose of this valuation literature review, we aimed to map the valuation studies onto these twelve habitats where possible, but for pragmatic reasons as explained below, we added extra categories: estuaries (including fjords and bays), intertidal wetlands, coral reefs (tropical) and mangroves.

Our literature review reveals that the habitats covered by valuation studies are sometimes less precise or more pragmatically defined than the sediment-based marine habitats of the UK NEA or the EUNIS classification which are depth and salinity based (see **Table 4.9.**). For example, estuaries may encompass different intertidal (e.g. mud flats) and shallow subtidal areas (e.g. seagrass beds, kelp forest), as well as coastal margins (e.g. salt marshes) (Moss, 2008). Because of this habitat complexity, valuation studies often do not or cannot assign ecosystem goods and benefits to specific habitat types within an estuary and broadly label the study area as an estuary. Similarly, valuation studies report to provide values for '(intertidal) wetlands', which may encompass other habitats, such as marshes and mudflats. Other valuation studies do not provide sufficient detail about the study area to assign values to either intertidal or subtidal areas, littoral or sublittoral areas, or to specific areas in the coast or open ocean. Valuation studies included in the 'coastal shelf' category may also include different coastal and marine habitat types, depending on the study area. Where possible, we allocated these studies to specific habitat types, but when this was impossible the study was included in the (therefore broad) coastal shelf category. In addition, we assigned economic values to the coastal shelf if the fisheries pertained to Exclusive economic zones (EEZ). These EEZ boundaries are political boundaries, however, and do not coincide with the depth-based boundary between open ocean and coastal shelf as in the UK NEA 2011. Mapping valuation studies onto the UK NEA 2011 or EUNIS categories was considered to be impractical or impossible from a valuation perspective, and not imperative from a modelling perspective.

Table 4.9. Coastal and marine habitat classification in UK NEA 2011 and UK NEAFO.

	UK NEA 2011	UK NEAFO Literature review	EUNIS*
Coastal margins	Sand dunes, incl. sand beaches	Dunes	B1.3 - B1.8
		Beaches	B1.1, B1.2
	Shingle, incl. shingle beaches		B2
	Machair	Machair	B1.9
	Sea Cliffs (hard/soft), incl. small islands	Cliffs and small islands	B3
	Salt Marsh	Salt and tidal marshes	A2.5
	Coastal Lagoons	Lagoons	X02, X03
Marine	Intertidal rock and biogenic reefs	Cold water corals	A5.64
		Coral reefs (tropical)	-
	Intertidal Sediments (salt marshes, muds, sea grass)	Mudflats	A2.2, A2.3
		Intertidal wetlands	A2.5
		Mangroves	-
		Sea grass beds	A2.6
	Shallow subtidal sediment (sea grass, kelp, maerl)		A5.53, X32
		Kelp forest	A3.2, X32
		Estuaries, fjords and bays	X01
	Subtidal Rock	Rocky bottom	A1, A2.1, X31
	Shelf subtidal sediment	Coastal shelf (EEZ)	-
	Deep-sea habitats (below 200 m)	Open ocean (beyond EEZ)	-

* A refers to marine habitats; B to coastal habitats; X to habitat complexes (mixed types)

Tropical coral reefs and mangroves are located in (sub) tropical climes and therefore not of immediate importance to the UK NEA, but nevertheless they were included to give a global perspective and overview of the valuation literature.

4.6.2.3 Valuation methods

A number of valuation methods have been developed to estimate the value of any good. They range from adjusted market prices, through productivity effect (production function) methods and revealed preference (based on consumer actions) to survey-based expressed preference methods. At the simplest level, market prices, for example, can be used to estimate part of the value of improved water quality by quantifying the increased value of commercial fish catches.

Different economic valuation techniques will be appropriate for different goods and benefits (see **Table 4.10.**), but it will not be possible to place meaningful monetary values on all the benefits (and some of the costs) of outputs from the coastal and marine zone. In particular the symbolic and cultural values assigned to some coastal and marine features and land/seascapes lie outside the monetary calculus and

are conditioned by social preferences and norms (shared values) arrived at, over time, through various forms of information transmission, art, literature etc.

Table 4.10. gives an overview of the different valuation methods that can be used for valuation of coastal and marine ecosystem goods and benefits, including the human welfare measure they are based on, and the goods and benefits they can be applied to. One important issue to consider when using existing primary valuation studies and comparing their results, is related to the fact that studies may use different valuation methods. Valuation methods do not necessarily address similar constructs of welfare, for example, producer surplus versus consumer surplus, or net versus gross revenues. Different welfare constructs, strictly speaking, cannot be added up or compared – they are different types of estimates.

Table 4.10. Overview of environmental valuation methods for estimating individual WTP. Source: Based on Brander *et al.* (2006) and Turner *et al.* (2010).

Method	Short description	Limitations	Welfare measure	Good/benefit
Travel cost method	Recreational benefits. Indirect method. Estimate demand (WTP) using travel costs to visit site.	Large data requirements, complex when trips are multipurpose, only for use values	Consumer surplus	Recreational angling, beach visits, diving, other recreational activities
Hedonic pricing method	Amenity benefits. Indirect method. Estimate WTP using price differentials and characteristics of related products.	Large data requirements, sensitive to model specification, only for use values	Consumer surplus	Amenity - property (housing, hotels, land)
Contingent valuation	All goods, also non-use values. Direct survey-based method. Hypothetical questions to obtain WTP.	Time and cost intensive, biases related to non-compensatory behaviour, constructed preferences and framing effects.	Compensating or equivalent surplus	Appreciation of culture, heritage, recreation, landscape, biodiversity. Bundle of services
Choice experiments	All goods, direct method, also non-use values. Hypothetical questions to obtain WTP	As for CVM, and: greater cognitive burden, and associated learning and fatigue biases.	Compensating or equivalent surplus	Appreciation of culture, heritage, recreation, landscape, biodiversity. Bundle of services
Net Factor income	Assign value as revenue of an associated product net of costs of other inputs	Only applicable to marketed goods, tends to overestimate values.	Producer surplus	Commercial fishing, aquaculture and other products, tourism

Method	Short description	Limitations	Welfare measure	Good/benefit
Production function (dose-response)	Estimate value as an input in production. Trace impact of physical change of an ES on human welfare.	Data on change in service and impact is often unavailable, only applicable to use values.	Producer and consumer surplus	Sea defence, erosion control, healthy climate, fishing and other products
Replacement cost	Costs of replacing the function with an alternative (manmade) technology or restoration of the ecosystem	Only applicable to use values, tends to overestimate value		Sea defence, erosion control, healthy climate
Defensive / preventive expenditure method, avoided damage costs	Costs and expenditures incurred in avoiding damages of reduced environmental functionality	Only applicable to use values, substitutability issues, typically lower bound estimate, problematic when goods are produced jointly		Sea defence, erosion control, landscape and biodiversity, health, etc
Market prices	Accounting procedure applicable to market traded goods.	Assumes perfect markets, only possible for private goods, lower bound estimates.	Total market revenue of goods or services	Commercial fishing, aquaculture, harvesting of products

Another word of caution concerns the use of cost-based approaches. The replacement cost approach looks at the costs of replacing an ecosystem service by a manmade alternative (either a technology or re-created habitat). This approach assumes that if society is willing to pay these replacement costs, the value of the ecosystem benefit must be *at least* that amount, and they may be higher. As such, these cost-based estimates provide a lower bound estimate of the societal value of ecosystem goods and benefits.

By their nature, valuation methods differ in terms of the unit used to represent value estimates: some methods result in a value per unit area or physical (qualitative or quantitative) change in ecosystem delivery, other studies will provide a value per household or individual for a (small set of) change(s) in area or ecosystem services provision. Some studies provide total values for an entire habitat area, for instance, the total value of fisheries along the UK coast.

Marginal economic values, relating to an incremental change in ecosystem service provision, are grounded in economic theory and used in the assessment of changes in welfare over time. In order to scope the uncertain future outcomes, scenario analysis is often deployed in which a change from the baseline to a future state of the world is considered. For scenario analyses working with land use change maps, marginal values per unit of area are most practical. However, marginal values may not be proportional to biophysical unit changes (quantity, quality, area). There is no a priori expectation about the relationship. In general, prices (values) are expected to increase as supply decreases, but some ecosystems will have thresholds below which no services are provided. Therefore, as an example, losing the last ha of wetland will have a different value than losing the first ha, and average values per ha (total

benefit flow divided by total area) should be used with caution (see Brander *et al.*, 2012 for a discussion). Per ha values also do not account for changes in value related to changes in depth.

Values expressed in different units will also require a different aggregation process; some values may be aggregated over the relevant area, whereas others are to be aggregated over the relevant population. When marginal values are not proportional to unit area, aggregation errors may arise.

Whatever methodology is used to conduct the assessment, all results should be subjected to a rigorous uncertainty and sensitivity analysis. Uncertainty is present at all stages of the assessment process, whether it be uncertainty about the magnitude of physical impacts and their geographical and temporal distribution, or uncertainty over the value of changes in ecosystem benefits and goods. Sensitivity analysis allows this uncertainty to be explored in a constructive manner and can be used to identify the parameters of the system which are particularly subject to uncertainty and that have a significant impact on the overall outcome of the assessment.

4.6.2.4 Financial analysis

A number of reports (Posford Duvivier Environment, 1996; Pugh & Skinner, 2002; Pugh, 2008; Saunders *et al.,* 2010; UKMMAS, 2010) review the financial values (e.g. in terms of gross value (GVA) added to the UK economy) of marine-dependent industries, including fisheries and tourism. Where possible, this report aims to provide an overview of *economic* values (related to human welfare) rather than *financial* values. Financial values will typically diverge from economic values when so-called public goods are involved which lack private ownership, or when the full costs of production and consumption (especially environmental impact costs) are not readily included in the pricing process (see **Section 4.2.8.3**). For many ecosystem goods and benefits there are no markets available, or the full cost of supply are not reflected in financial measures. Nevertheless, financial data does provide a useful, albeit restricted, perspective on ecosystem service losses/gains, and often is the only data available.

Pugh (2008) gives an overview of the contribution of the marine activities to the UK economy in terms of GDP and employment, based on official Government statistics. This Crown Report provides an update of the 2002 report by Pugh and Skinner. The Productive Seas Feeder Report (UKMMAS, 2010), providing an evidence base for the Charting Progress 2 (state of the UK seas) report, summarises the evidence of productive uses of UK seas. It looks at various industries that use the marine environment to produce goods that are marketed. Saunders *et al.* (2010) build on this report to develop indicators that can be used to assess changes in coastal and marine scenario analysis. These overviews include both biotic and abiotic industries, such as energy, shipping and marine aggregates.

Table 4.11. summarises the findings of these reports for the eight ecosystem goods and benefits categories that we have identified. The table includes estimates of GVA. Estimates of GVA exclude potential externalities of these sectors. With respect to the value of fisheries, the GVA estimates provided here include secondary activities such as the processing of shellfish and finfish. It shows that many of the ecosystem goods and benefits fall beyond more traditional measures of economics such as GDP, yet they contribute to human well-being.

The aim of this report is therefore to see if there are existing valuation studies that can be used to assess the values of these ecosystem goods and benefits in benefit transfer (see **Section 4.6.4.4**). Benefit transfer (BT) is the use of research results from pre-existing primary studies at one or more 'study sites' or contexts to predict welfare estimates such as willingness to pay or related information for other,

typically unstudied 'policy sites' or contexts. In addition, we aim to focus on the economic values rather than financial values, and assess the non-market values, for instance, for recreation and leisure.

Table 4.11. Overview of financial value estimates of ecosystem services goods and benefits.

	GVA* (Pugh, 2008) (in £million)	GVA* (UKMMAS, 2010) (in £million)
Aquaculture, fisheries and processing	808	887
Sea defence	n.a.	n.a.
Erosion control	n.a.	n.a.
Healthy climate	n.a.	n.a.
Tourism and recreation	3,326	4,550
Property-related amenity	n.a.	n.a.
Education and research	478	171
Biodiversity, species, habitat conservation	n.a.	n.a.

*GVA = Gross Value Added. n.a.: not available.

4.6.3 Methods and design: selection and quality criteria

This report provides an overview of valuation studies addressing the monetary value of market and non-market ecosystem goods and benefits provided by coastal and marine ecosystem services. The overview covers articles providing primary valuation studies published since 2000 in academic journals and book chapters and have undergone peer-review. Papers published in grey literature (consultancy and non-governmental organisation (NGO) reports, working papers) or before 2000 have been excluded. This is to some extent a subjective judgement, but valuation estimates are subject to serious spatial and temporal bias constraints and in the latter context a period of more than a decade or so was thought to be a prudent limit. Peer-revision is taken as a quality assessment of the analysis, whereas the quality of grey literature can often not be tested as these studies provide little information about the methods of data collection, analysis and tests, the valuation process and political context.

After identifying the main gaps in the literature (important goods and benefits for which no primary studies were available), we decided to mention some available primary UK case studies published between 1990-2000, as well as some high quality consultancy reports and international studies that may be used (with necessary caution) to fill the gaps.

The selection process was based on web-searches in Science Direct and Google Scholar using the key-words 'ecosystem services', '(economic) valuation', 'coastal', 'marine', in various combinations. Secondly, primary studies referenced in the selected studies, available meta-analyses, the UK NEA 2011 report or other review papers (e.g. Beaumont *et al.*, 2008; 2010) were included. Finally, we performed a more targeted search on specific journals (e.g. Ecological Economics, Environmental and Resource Economics, Journal of Environmental Economics and Management, Land Economics, Marine Resources Economics, Ocean and Coastal Management) and authors to complete the list. The selection processes is limited to data available up to 1 May 2013; articles published after this date are not included in the overview.

From each selected study, we extracted information on the authors, year of publication, continent and country of the case study, valuation method, habitat type and ecosystem goods and services under consideration. For the GB-based studies, we also extracted value estimates. These estimates were

converted to 2012 prices and expressed in GBP. For each of the GB studies, we provide a short description of the study, covering the valuation method, main results, main limitations, suitability for benefit transfer and scenario analysis.

We then reviewed the valuation studies based on a number of criteria that qualify studies for benefit transfer purposes (Brouwer, 2000). Firstly, the studies must use adequate data, sound economic methods, correct empirical techniques, and a model or WTP function with valid explanatory variables. Secondly, the population of beneficiaries must be described, including the distribution and characteristics. Thirdly, the ecosystem good or benefit and the (change in) provision level must be specified. Finally, the site characteristics must be described, because similarity of sites is important in benefit transfer. This review focuses mainly on the first and third set of indicators; populations can be based on secondary (census) data, and the studies usually include a site name and description. It should be noted that there is a large variation across studies due to differences in the applied valuation techniques. Therefore, generic quality criteria are difficult to apply. We judged each study in terms of its own method as systematically as possible (see also the systematic review protocol of the journal Environmental Evidence).

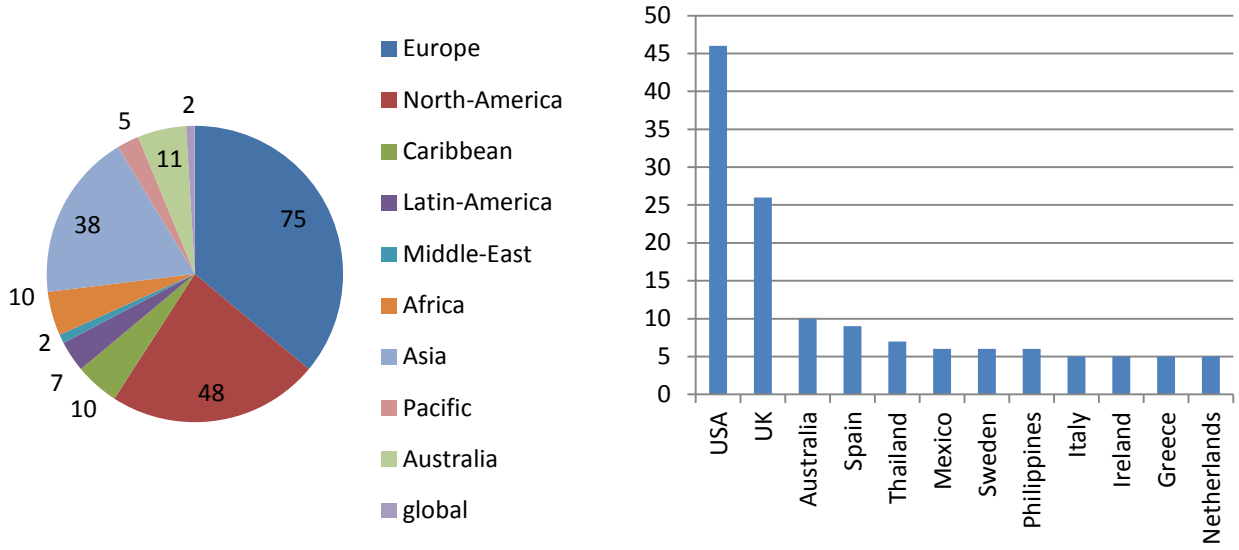
In **Section 4.6.4.3**, we provide a short description of the available values for each of the groups of ecosystem goods and services. It should be noted that value estimates are often dependent on the nature of the study, i.e. the policy context, the valuation method, the sample and the survey design. Results have to be interpreted with such study effects in mind.

4.6.4 Results

4.6.4.1 Descriptive statistics

This section gives some descriptive statistics related to the compiled dataset of coastal and marine economic valuation studies. The selection process resulted in 208 primary valuation studies, including 26 UK studies, published since 2000 in peer-reviewed academic journals and books. In addition, we identified nine relevant meta-analyses.

There is no strong positive trend in the number of publications in the academic literature with monetary value estimates of coastal and marine ecosystem goods and benefits since 2000. Stated preference (SP) methods, including contingent valuation (CV) and choice experiments (CEs), are used most frequently, mainly to assess recreational and biodiversity values, followed by travel cost (TC) assessments for recreational values and estimation of gross or net revenues to assess benefits of raw materials (mainly fishing). The majority of studies address case study areas in Europe and North-America (see **Figure 4.43.**). The North-American studies are mostly for the USA. A third of the European case studies are for the UK. However, it may be that the number of UK-studies is biased upwards due to our focus on UK-based valuation evidence.



(a) (b)

Figure 4.43. a and b: Number of studies per continent (a) and per country (b). Figure 4.43b only includes countries for which five or more studies are available.

Globally, 'tourism and recreation' is the most frequently valued ecosystem benefit, followed by biodiversity and habitat benefits (see **Figure 4.44.**). This corresponds to the high numbers of SP and TC studies. Most of the tourism studies are for tropical coral reefs, beaches and coastal areas. There are very few valuation studies for ecosystem benefits related to prevention of coastal erosion, and education and research. Surprisingly, only a small number of value estimates are available for the carbon sequestration potential of coastal and marine habitats.

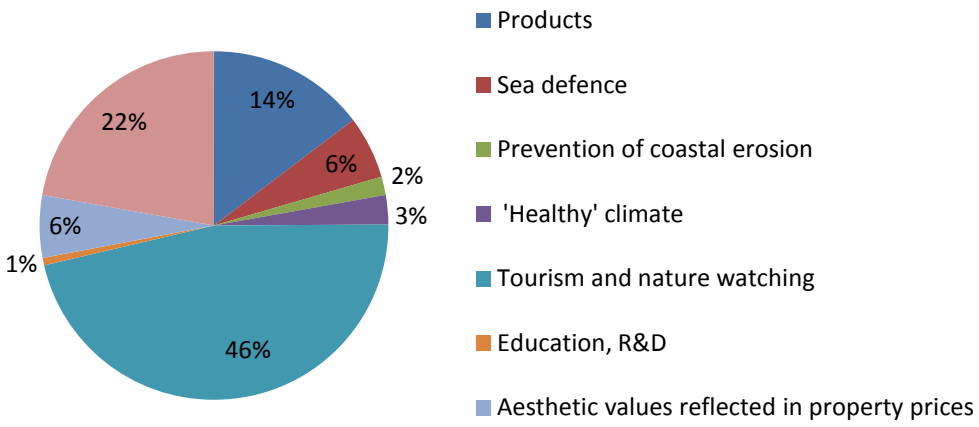


Figure 4.44. Percentage of studies for each category of ecosystem goods and services. The percentages refer to the number of studies (n=208). Some studies may provide multiple values for different ecosystem goods and services.

The distribution of studies across the different habitats shows that there are no studies for machair and only one study for cliffs and small islands, one for rocky bottoms and one for cold water corals. Dunes, coastal lagoons, mudflats, kelp forests, rocky bottoms, open oceans, and cold water corals have also

received very little attention in the valuation literature. The ecosystem goods and benefits provided by beaches, tropical coral reefs and the coastal shelf have been most frequently valued in the literature.

4.6.4.2 GB-based studies

The 25 primary GB valuation studies cover various habitats and goods and benefits. Recreational values are most frequently provided in the literature. The studies cover:

- fisheries values from the coastal shelf/EEZ and salt marshes;
- sea defence benefits provided by dunes, shingle beaches, salt marshes and mudflats;
- prevention of coastal erosion related losses by shingle beaches;
- 'healthy' climate benefits provided by salt marshes, mudflats, dunes, seagrasses, kelp forests and the coastal shelf[35];
- recreational values by salt marshes, mudflats, beaches, dunes, small islands and coastal areas; and
- spiritual and cultural wellbeing and aesthetic benefits of wild species and seascapes for salt marshes, and generic non-use values for wetlands and the coastal shelf.

Two studies describe large scale analyses in which coastal areas are included, but they are not habitat specific. Sen *et al.* (2013) provide an analysis which covers all terrestrial habitats and also includes coastal areas and beaches. Mourato *et al.* (2010) assess the effect of coastline on housing prices. As **Table 4.12.** shows, there are no UK valuation studies for a number of the habitats and ecosystem goods and benefits, published in the academic literature since 2000. No habitat specific values are available for any of the goods and benefits provided by machair, estuaries, rocky bottoms, cold water corals and open oceans.[36] There are no habitat-specific value estimates at all for amenity effects on housing prices (property values), and only one study on benefits of prevention of coastal erosion. Furthermore, there are many habitats for which only a couple of goods and benefits have been assessed.

[35] Mangi *et al.* (2011) provide carbon sequestration values for seagrasses, kelp forest and the coastal shelf, but this study does not provide sufficient detail about the value estimates.

[36] Tropical coral reefs and mangroves are of little importance to GB and therefore not included in **Table 6.4**.

Table 4.12. Overview of number of UK studies for each combination of habitat and ecosystem service. The numbers refer to the number of studies that provide at least one value for the ecosystem service in a particular habitat type since 2000. Yellow indicates services for which one to four studies are available, and green indicates that five or more studies are available for a service in a habitat type. The recreation study by Sen *et al.* (2013) and the hedonic pricing study by Mourato *et al.* (2010) are not included in this overview because they cannot be assigned to habitats. The study by Mangi *et al.* (2011) is excluded because the study does not provide sufficient information to convert the value estimates into 2012 prices for annual flows.

	Pro-ducts	Sea defence	Erosion preven-tion	Healthy climate	Tourism and nature watching	Education research	Aesthetic: property	Spiritual/ aesthetic: wild species, seascapes
Dunes	0	2	0	1	1	0	0	1
Beaches	0	1	1	0	4	0	0	0
Cliffs,small isl.	0	0	0	0	1	0	0	0
Machair	0	0	0	0	0	0	0	0
Lagoons	0	0	0	0	0	0	0	0
Marshes	1	4	0	4	1	0	0	1
Mudflats	0	2	0	2	1	0	0	0
Inter. wetland	0	0	0	0	0	0	0	1
Seagrass beds	0	0	0	0	0	0	0	0
Kelp forest	0	0	0	0	0	0	0	0
Estuaries	0	0	0	0	0	0	0	0
Coral reefs	0	0	0	0	0	0	0	0
Rocky bottom	0	0	0	0	0	0	0	0
Coastal shelf	3	0	0	1	5	0	0	2
Open ocean	0	0	0	0	0	0	0	0

4.6.4.3 GB studies per ecosystem goods and benefits group

Table 4.13. provides an overview of the available value estimates. Unless stated otherwise, value estimates in this section are expressed in £, 2012 prices. Original values reported in the original studies have been corrected for inflation, using the National Accounts figures from ONS (last updated 27 March 2013).

Table 4.13. Overview of UK valuation studies published since 2000 in the academic, peer-reviewed literature.

Ecosystem service	Habitat	Case study and reference	Valuation method	Value as reported in study (£/yr)	Value in 2012 prices (£/yr unless stated otherwise)
Products: Fisheries (nursery)	Salt marshes	contribution to commercial fishing of Blackwater realignment (Luisetti *et al.,* 2011)	Market prices	7.43-11.55/ha (2007) (after 5 years)	8.27-12.86/ha (after 5 years)
Products: Fisheries	UK coast/ open sea	Cod fisheries in North Sea (Crilly & Esteban, 2013)	Gross value	12M in 3 years (2006-2008)	4.4M
		Fisheries and (shell)fish farming (Austen *et al.*, 2010 ; Beaumont *et al.*, 2010)	Gross value	Fisheries: 596M (2010) Fish farms: 327M (2007) Shellfish farms: 23M (2007)	Fisheries: 619M Fish farms: 364M Shellfish farms: 26M
Healthy climate	Dunes	(Beaumont *et al.*, 2010)	Abatement costs (DECC)	32–242/ha (2010)	33-251/ha
	Salt marsh & mudflats	(Andrews *et al.*, 2006)	SCC	12/ha (2004-05)	14/ha
		(Shepherd *et al.*, 2007)	SCC	11-45/ha (2004-05)	13-53/ha
	Salt marshes	(Luisetti *et al.,* 2011)	Various prices (4-230/tC)	1-770/ha (2007)	1-865/ha
		(Beaumont *et al.*, 2010)	Abatement costs (DECC)	61–622/ha (2010)	63-646/ha
	Sea grasses	(Luisetti *et al.,* 2013a)	Various prices: DECC (54), (SCC (3.33-233/tC))	-	103/ha (6.36-445/ha)
	Coastal shelf	(Beaumont *et al.*, 2010)	Abatement costs (DECC)	6.74 billion	Total: 7 billion (+/-50%)
Coastal erosion prevention	Shingle bank (beach)	Recreational values of freshwater marshes protected by shingle bank (Bateman *et al.,* 2001)	TC CV	TC: 50/hh/visit (2000) CV: 1.58-62.08/hh (2000)	TC: 66/hh/visit CV: 2-81/hh

Ecosystem service	Habitat	Case study and reference	Valuation method	Value as reported in study (£/yr)	Value in 2012 prices (£/yr unless stated otherwise)
Sea defence	Dunes	(Beaumont *et al.*, 2010)	Replacement costs	England: 174-520M (2010); Wales: 54Bn (2010)	England: 181- 540M; Wales: 56Bn
		(Van der Meulen *et al.*, 2008)	Management costs	285- 1800 €/ha (2001)	309-1949/ha
	Shingle beaches	(Beaumont *et al.*, 2010)	Replacement costs	England: 0.79Bn (2010)	England: 0.82Bn
	Salt marshes and mudflats	(Andrews *et al.*, 2006)	Cost based (replacement/ avoided)	Capital costs: 878,159/km; opportunity costs: 2282-2,576/ha; savings on investments (one off): 668,441/km; maintenance costs savings: 3,170-3,560/km	Capital costs: 1,033,420/km; opportunity costs: 2,685-3,031/ha; savings on investments (one off): 786,623/km; maintenance costs savings: 3,730-4,189/km
		(Shepherd *et al.*, 2007)	Cost based (avoided costs)	maintenance costs savings: 4,206/km; (5,546->1,340)	maintenance costs savings: 4,950/km; (6,527->1,577)
	Salt marshes	(Beaumont *et al.,* 2010)	Net replacement costs Replacement costs	England: 2.17Bn (2010) 1,500-3,500/m wall (1994) 2,600-4,600/m wall (1994) 3.7-6.6Bn (1994)	England: 2.25Bn; 2,225-5,191/m wall; 3,856-6,822/m wall 5.5-9.7 Bn
Tourism and nature watching	All	Coastal recreation (Sen *et al.*, in press)	Meta-analysis	3.96/trip (2011) England: 38 M	4/trip England: 39M
	Beaches	Norfolk EC Bathing Water Directive (Georgiou *et al.*, 2000)	CV	35.73/hh (1997)	49/hh
		Norfolk beach replenishment (Bateman *et al.*, 2001)	CV	25.84 - 31.62/hh (local – holiday) Total: 741,517 (2000)	34-41/hh (local – holiday) Total: 971,640
		Coastal water quality in Scotland (Hanley *et al.*, 2003)	TC, Contingent behaviour	0.48/trip (5.81/pp/yr) (1999); Total 1.25M	0.63/trip (7.66/pp) Total: 1.65M
		Beach protection (Christie	CE	Beach safety: 33.4/hh	Beach safety: 38/hh;

Ecosystem service	Habitat	Case study and reference	Valuation method	Value as reported in study (£/yr)	Value in 2012 prices (£/yr unless stated otherwise)
		& Gibbons, 2011)		(2006); Surfing conditions: 14.5/hh (2006)	Surfing conditions: 16.5/hh
	Cliffs, small islands	Lundy Island Marine Nature Reserve (Chae *et al.*, 2012)	TC	359-574/trip (2005)	420- 672/trip
	Salt marshes	Blackwater managed realignment (Luisetti *et al.*, 2011)	CE	Total for case study area: 4,429-8,348 (2006) Access: 4.31/hh (2006) Area (ln (ha)): 1.11/hh (2006)	Total for case study area: 5,041-9,501 Access: 4.91/hh Access (ln(ha)): 1.36/hh
	Coastal shelf	Seal conservation in England (Bosetti & Pearce, 2003)	CV	8-9/view (seal conservation) (2000)	10-12/view
		Whale-tourism in West-Scotland (Parsons *et al.*, 2003)	Gross value	Whale-tourism: 1.77 M (2000); Total: 6 M (2000)	Whale-tourism: 2.3 M Total: 7.9 M
		Sea angling in England (Lawrence, 2005)	CE	Per day-trip: 5.60-12.45 (2004)	Per day-trip: 6.72-14.93
		Lyme Bay, England (Rees *et al.*, 2010)	Gross value	18.3M (angling, diving, wildlife watching: 13.7, 1, 3.5) (2008)	19.8M (angling, diving, wildlife watching: 14.8; 1.1; 3.8 M)
		Biodiversity related recreation in Wales (Ruiz Frau *et al.*, 2012)	Gross value (financial revenues)	Diving: 7.8M; kayaking: 2.5M; boating: 13.4M (2008); seabird watching: 3.7M (2009)	Diving: 8.4M; kayaking: 2.7M; boating: 14.5M; seabird watching: 3.9M

Ecosystem service	Habitat	Case study and reference	Valuation method	Value as reported in study (£/yr)	Value in 2012 prices (£/yr unless stated otherwise)
Spiritual and cultural wellbeing and aesthetic benefits of wild species and seascapes	Salt marshes	Blackwater managed realignment (Luisetti *et al.,* 2011)	CE	Additional bird species: 1.84–3.57/hh (2006)	Additional bird species: 2.09-4.06/hh
	Intertidal wetlands	Otter and bird protection (Birol & Cox, 2007)	CE	Otter hold creation: 31.6/pp; Protecting birds: 1.2/pp	Otter hold creation: 37.19/pp; Protection birds: 1.41/pp
	Coastal shelf	MPAs in the UK (McVittie & Moran, 2010)	CE	Halting loss of biodiversity and ES: England: 69/hh; Wales: 107/hh, Scotland: 21/hh, NI: 34/hh (2008) Increasing biodiversity: England: 69/hh, Wales: 61/hh, Scotland: 24/hh, NI: 38/hh (2008).	Halting loss of biodiversity and ES: England: 75/hh; Wales: 116/hh; Scotland: 23/hh, NI: 37/hh. Increasing biodiversity: England: 75/hh, Wales: 66/hh, Scotland: 26/hh, NI: 41/hh.
		Marine species conservation (Ressurreicao *et al.,* 2011; 2012)	CE		Mammals: 43-49/hh; Birds: 39-44/hh; Fish 38-43/hh; Invertebrates: 36-41/hh; Algae: 46-53/hh. All one-off payments.
		Seal conservation in England (Bosetti & Pearce, 2003)	CV	Non-use: 526,000 (2000)	Total: 689,239

Note: M: million.

Products

The products that people derive from coastal and marine ecosystems, include not only aquaculture and fisheries for food provision, but also bait, fish feed, fertiliser, ornaments and medicines.[37] Most studies on the benefits of products cover commercial fisheries. Four studies provide primary data for the UK. Luisetti *et al.* (2011) estimate the contribution of created **salt marshes** (through coastal realignment schemes) to commercial fisheries for salt marsh creation in Blackwater estuary. The fish production is based on estimates of juvenile bass abundance per 0.1ha, and combined with average survival rates of fish up to commercial sizes. Based on local market prices, the estimated economic contribution to the local inshore fishery is £11.55/ha of salt marsh, starting 5 years after the salt marsh is re-created. According to Luisetti *et al.* (2013b), fish production functions are highly site-specific and transferring the function from the Blackwater site to another salt marsh would not be reliable.

The three other studies look at **coastal shelf** areas. Between 1993 and 1995, on average 37 species contributed more than €1million per year[38] to the value of the fisheries sector of France and the UK (Mardle *et al.*, 2002), with highest revenues for the UK fleet from scallop, sole, mackerel and edible and spider crabs. The UK is also responsible for 38% of the annual North Sea cod catches for the past decade (Crilly & Esteban, 2013), with an annual value of approximately £4.4million. Chapter 12 of the UK NEA 2011 (Austen *et al.,* 2010) provides a detailed overview of landing data for the UK and the financial value of these landings. In 2008, the gross value was approximately £619million (Beaumont *et al.*, 2010), with a higher value of shellfish than of demersal and pelagic species.

These annual values cannot be split into values per unit area without data on vessels activities across the coastal waters. Beaumont *et al.* (2010) discuss other limitations of the use these estimates in future projections and scenarios, caused by problems with sustainable harvesting levels. Fisheries also have other negative externalities, which are not reflected in market prices. Crilly and Esteban (2013) argue that per tonne of cod, the societal value of trawlers is negative, if the value of discards and GHG emission costs are subtracted, and the sector also receive fuel and direct subsidies. The societal value depends on gear; gillnet fisheries, although much smaller in terms of their share in the value of landings, have a positive societal value, mainly because their discards and fuel consumption are much lower per tonne of cod, compared with trawlers.

Apart from fisheries, other revenues are derived from aquaculture and seaweed extraction. In financial terms, finfish and shellfish farming in the UK generated a turnover of £364million (mainly generated by Scottish salmon farms) and £26million respectively in 2007 (Austen *et al.*, 2010). It should be noted that these turnover rates may not reflect potential negative impacts of aquaculture on the environment and therefore do not reflect the full societal welfare impacts.

No UK studies on products are available other than those on (shell-) fisheries and aquaculture. Part of the extraction of seaweed in the UK is for use in medicine production (Beaumont *et al.*, 2008), but no estimates are available that allow a breakdown of seaweed revenues into various uses including their pharmaceutical value. Moreover, there are no studies for beaches, dunes, sea cliffs and small islands, cold water corals, mudflats, and rocky bottoms.

[37] Note that recreational extraction of food and other products are included in the section on recreation.

[38] Value could not be converted to £, 2012 due to a lack of information in the original study.

'Healthy' climate

Estimating new monetary values for the range of benefits that people derive from a 'healthy' local and global climate, or the assessment of the damages that can be avoided by emitting less GHG, often falls beyond the scope of valuation studies that assess these benefits provided by coastal and marine ecosystems. Typically, valuation studies use existing estimates of carbon sequestration rates of coastal and marine ecosystems and apply these to their case study area, combined with a price per tonne of carbon. Prices and values per tonne of carbon can be determined in different ways (see Luisetti *et al.*, 2011). Since climate change is a global phenomenon, and it does not matter for climate change where carbon is emitted or sequestered, the price per tonne of carbon is uniform. But different valuation methods can be used: DECC prescribes prices based on the abatement cost method (looking at the cost of measures to reduce emissions), various studies use Social Cost of Carbon (SCC) estimates (that capture welfare changes associated with the impact of climate change), and market prices can also be used (although these markets are far from perfect in many ways, and crashed in April 2013). We include studies that provide monetary estimates. **Appendix 4.6** reviews the available non-economic literature on carbon burial by coastal and marine ecosystems.

Jones *et al.* (2008) provide a study on **dunes** based on a UK site and this estimate has been used in the UK NEA 2011. Carbon accumulation rates are estimated to be 0.582±0.262tC/ha/yr in the dry dunes and 0.730±0.221tC/ha/yr in the wet dune habitats. Beaumont *et al.* (2010) use these estimates to estimate the value for the provision of C sequestration by dunes habitats using the 2010 DECC CO_2 price (£51.6+/-50% in 2010). The resulting values range from £33 to £251/ha/yr (in 2012 prices).

Andrews *et al.* (2006) estimate the value of annual carbon storage in Humber **mudflat and salt marsh** sediments, using the average concentrations of particulate C, N and P from Andrews *et al.* (2000) and Jickells *et al.* (2003). They calculate that an extension of 7494ha of intertidal area through managed realignment would annually bury about 3597t of organic carbon (0.48tC/ha/yr), resulting in climate benefits of £14/ha/yr, valued against a SCC estimate. Shepherd *et al.* (2007) also use Andrews *et al.* (2000) and Jickells *et al.* (2003) to estimate the carbon sequestration benefits of managed realignment in the Blackwater Estuary. They estimate that the creation of 2950ha of **salt marsh** and 2370 ha of **mudflat** would result in additional carbon storage of 2354.4tC/yr (0.44tC/ha/yr) using a sedimentation rate of 1.5mm/yr. With a rate of 6mm/yr, the additional storage would be 9417.7tC/yr (1.21tC/ha/yr), associated with benefits of £13-53/ha/yr, using a SCC estimate. Luisetti *et al.* (2011) value the carbon storage capacity by **salt marsh** re-creation projects in the range of £1-865/ha/yr, depending on the price per tonne of carbon used (market prices, SCC estimates) and the sedimentation rate, which is expected to vary between 0.266tC/ha/yr and 3.347tC/ha/yr (Andrews *et al.,*2000; Adams *et al.,* 2012). Beaumont *et al.* (2010) estimate that the value for carbon sequestration by **salt marshes** ranges from £63-£646/ha/yr. This estimate is based on a long term soil carbon sequestration rate of salt marshes from Cannell *et al.* (1999) of 0.64-2.19 tC/ha/yr (2.35 – 8.04 tCO_2/ha/yr), combined with the DECC price for non-traded carbon.

Luisetti *et al.* (2013a) published estimates on the carbon sequestration of **sea grass** (*Zostera marina* species), of which the UK has 4887 ha. Using the sequestration rate of 1.91 tCO2/ha/yr from this study, the value of the benefits of carbon sequestration can be derived and is estimated at £103/ha/yr, valued against the central, non-traded DECC price for 2012. Depending on the price of carbon used in Luisetti *et al.* (2013a), this value may vary between £14 and £894/ha/yr. The low estimate is based on Pearce (2003), while the high price is based on Stern (2007) who applies a relatively low discount rate.

Beaumont *et al.* (2010) also estimate that in 2004, the value of carbon sequestration in marine habitat by phytoplankton based on primary production was £7 billion/yr +/- 50% (in 2012 prices, based on 2010 DECC non-traded carbon prices, given the extent in salt marsh in 2004). This figure reflects productivity in UK (**coastal**) **shelf** seas where it is unlikely that this carbon will be transported to the deep ocean and sequestered permanently. Similarly, Mangi *et al.* (2011) use primary productivity estimates for carbon sequestration by phytoplankton in the coastal shelf near the Scilly Islands based on Smyth *et al.* (2005). However, these estimates are surrounded with great uncertainty regarding the net effect that primary production has on atmospheric carbon levels. Transport processes in the sea and ocean and to deep waters, amongst others, affect how much carbon is taken up by marine systems and how long this carbon is captured for (see Heckbert *et al.,* 2011). The base year of the prices used in Mangi *et al.* (2011) is also unclear, so the values of carbon sequestration in coastal shelf areas, as well as estimates for seagrasses and kelp forests (for which the carbon sequestration rates used in the analysis seem to be a factor 10 higher than in comparable carbon studies), cannot be expressed in 2012 prices and are therefore not included in **Table 4.13**.

In summary, the biophysical sequestration rates for dunes, sea grasses, salt marshes and mudflats used in the studies by Andrews *et al.* (2006), Shepherd *et al.* (2007), Luisetti *et al.* (2011, 2013a) and Beaumont *et al.* (2010) are comparable to other studies elsewhere. The biophysical and economic estimates are considered to be transferable across space and time (see Luisetti *et al.*, 2013b), and reflect marginal values that can be used for scenario analysis. No analyses are available for beaches, cliffs and small islands, machair, lagoons, intertidal wetlands, estuaries, cold water coral reefs, rocky bottoms and the open ocean. Beaumont *et al.* (2010) present values for the soil carbon stock in machair grasslands, but estimates for carbon sequestration (flow) by machair are unavailable. Additional research may also be needed for the valuation of carbon sequestration benefits provided by sea grasses, kelp forests and coastal shelf areas.

Prevention of coastal erosion

Natural habitats play an important role in coastal protection policies in the UK. In the period 1991 to 2009, around 1,180ha of land has been converted to (mostly) inter-tidal habitat for erosion control and sea defence purposes in managed realignment schemes (Roca *et al.*, 2010). The value of coastal erosion prevention includes avoided losses of property, agriculture, recreational uses.

The only study on benefits of coastal erosion prevention in the UK published since 2000 is reported in Bateman *et al.* (2001), addressing the recreational values of the freshwater Cley Marshes Natural Reserve in the UK that are protected by a **shingle** bank, using a combined TC-CV survey. The shingle bank that protects part of the Reserve from saltwater inundation cannot provide adequate production unless well maintained at a cost of £30,000-£50,000/yr. A combined TC-CV survey was executed to assess the recreational benefits of the Cley Marshes that could be saved by better management of the shingle bank. The results of the study show that the aggregate annual recreational benefits are around £1,970,000-£786,000, depending on the welfare estimate used (TC or CV) and the estimated number of visitors to the site. The TC estimates are about £66/hh/visit. The CV estimates vary between £2 and £81/hh/yr, although the authors argue that the lower bound estimate is probably too conservative. The results are based on a relatively small sample of n=160, and no details of the TC and CV WTP functions or analysis are given. Furthermore, the CV scenario does not quantify precisely what the effects of managed realignment on vegetation and birdlife would be – mainly because these were scientific unknowns. The usefulness of this study for benefit transfer (BT) may be limited. First of all, the erosion control benefits of the shingle bank are based on the value of the area it protects; hence, this study can only be applied to shingle beaches protecting freshwater marshes. Secondly, the transferability of TC

estimates has not been well investigated in the literature and would require information about number of visits per year. For transfer of the CV estimates, information about the relevant population of the study site would be needed.

An eftec report (eftec, 2010) suggests using BT to estimate the value of coastal erosion prevention and flood defence from the wetland meta-analysis estimates of Brander *et al.* (2008). For the UK NEA 2011, Morris and Camino (2010) perform a similar analysis for the storm buffering and flood control provided by coastal wetlands in the UK using the wetland meta-analysis by Brander *et al.* (2006). However, this and other meta-analyses often include a bundle of ecosystem goods and benefits and it is often impossible to extract values solely attributable to erosion prevention or sea defence. The well-known manual by Penning-Roswell *et al.* (2006) advocates the use of avoided damage cost approaches for estimating coastal erosion prevention values. However, we do not include any figures from the manual here, because no recent (post 1995) value estimates are available from this manual and the value of assets and the probability of flooding are site-specific.

Sea defence

Benefits of sea defence against sea flooding, storms and surges provided by coastal habitats include avoided damages to land, infrastructure, business, natural and historical environment, as well as the lower risk of life and the avoided costs of emergency rescue operations (e.g. evacuation) and the psychological effects of such events. The benefits of sea defence have been assessed for several ecosystems: marshes, mudflats, mangroves, beaches and dunes. Two existing meta-analyses have not found significantly higher values for storm protection provided by wetlands (Brander *et al.,* 2006) or lagoons (Enroljas & Boisson, 2010), but these results do not necessarily imply that these habitats do not provide sea defence or that the defence provided by these does not contribute to human welfare.

For the UK, there are a number of cost-based estimates available. Andrews *et al.* (2006) estimate that replacing hard defences by **salt marshes and mudflats** would save £786,623/km in terms of replacement costs of unsatisfactory hard defence and maintenance costs of £3,730–£4,189/km/yr. Where salt marshes complement existing hard defences, a net management cost saving of £4,950/km/yr (from £6,527 to £1,577/km/yr) can be achieved, according to Shepherd *et al.* (2007). These studies also provide more detail about the benefits and costs of managed realignment schemes. The one-off capital costs of realignment are estimated at £1,033,420/km, and the opportunity costs of agricultural land are between £2,685 and £3,031/ha. However, since salt marshes and mudflats also provide societal benefits through carbon sequestration, recreational opportunities and their nursery function, the overall cost-benefit ratio supports the implementation of this soft approach to coastal defence when viewed over >25year time scales.

The UK NEA 2011 report refers to estimates provided in King and Lester (1995) for sea defence benefits provided by a salt marsh in Essex, which allow for building lower man-made sea walls. The cost savings of salt marshes vary from £2,225-£5,191/m of wall for a 6m wide salt marsh beside the sea wall, to £3,856-£6,822/m of wall for an 80m wide marsh. Beaumont *et al.* (2010) use these estimates to calculate the costs of replacing UK **salt marsh** with man-made sea defences, by scaling by linear habitat length rather than area. The total value of these benefits amounts to approximately £5.5-£9.7billion, using DECC non-traded carbon prices. When using these figures for CBA, it should be noted that the sea defence provision also depends on the width of the habitat (Beaumont *et al.,* 2010), and the use of these relatively dated cost estimates (1995) may reduce the reliability of the value estimates.

Using net replacement costs estimates (i.e. the difference between sea wall construction costs and habitat maintenance costs) from the Environmental Agency (2007a), Beaumont *et al.* (2010) provide estimates related to the linear length of the habitat. They report a total replacement cost of £3.7billion, including for **shingle shores** (£0.82billion), **salt marshes** (£2.25billion) and **sand dunes** (£0.54billion) in England, but the authors list a number of limitations of this estimate, including inaccuracies in scaling by linear length which ignores variation in the altitude and value of protected land. An alternative approach, based on Pye *et al.* (2007), results in an estimated sea defence value for **dunes** of £56million in Wales and £181million in England, but this is a very conservative estimate as it only applies to dunes without any additional artificial defence structures near high value land.

The study by Van der Meulen *et al.* (2008) addresses the management costs of two **dune** sites on the Sefton Coast, one which is managed as a Nature Reserve and a busier one managed as a semi-park. Management costs vary between £309 and £1949/ha/yr. However, these costs are not only for sea defence, as these dunes are also managed for their recreational use and cultural/ spiritual/ aesthetic (biodiversity, non-use) benefits, but it is not possible to assign separate values to each of these benefits.

The main limitation of these cost-based estimates is that they do not reflect the value of the goods and benefits protected by ecosystem sea defence, including values of commercial and residential properties, agriculture and recreation. They are typically a lower bound estimate of society's willingness-to-pay. Moreover, the costs of managed realignment vary widely across sites (Tinch & Ledoux, 2006) and their transfer from the original 'study site' to the new, to-be-assessed 'policy sites' may therefore result in large errors in value estimation.

Pre-2000, Spurgeon (1998) reports on the costs of rehabilitation and creation of salt marshes in Essex whereby agricultural land along the coast is opened up to flooding. Cost-based estimates, in this case based on the engineering costs of securing existing defences further inland, are used as proxies of the benefits of salt marshes, resulting is estimates of $1,860(1993)/ha, $2,600(1995)/ha, and $43,000(1991)/ha. Estimates are based on a report by East Midlands Environmental Consultants. These costs are based on cost-effective approaches and do not include budgets for plant regrowth, which are assumed to happen naturally (but see Mossman *et al.*, 2012). Spurgeon (1998) also reports on the costs of creating marine lagoons in the UK for sea defence related purposes, noting an average cost of $7,000/ha (for moving earth, etc) based on projects in Norfolk and Cleveland. Estimates are based on personal communication with English Nature. For future value assessment, the use of more recent studies is recommended.

Tourism and nature watching

There are many international studies on the benefits of tourism for beaches, tropical coral reefs and coastal shelf areas, yet no value estimates for kelp forests, open oceans and cold coral reefs. In the UK, coastal and marine based tourism forms a considerable part of overall tourism, attracting national as well as international visitors. Activities include beach recreation, recreational angling, sailing, boating, nature-watching, diving, surfing and swimming. Saunders *et al.* (2010) and Chapter 12 of the UK NEA 2011 (Austen *et al.*, 2010) refer to financial estimates of some of these activities, based on government reports. Here, we give an overview of the UK based academic studies we found.

Sen *et al.* (2013) present the largest scale assessment of outdoor recreation in the UK, and the estimates are described into detail in the UK NEA 2011. Based on MENE data on visits to the natural environment

collected by Natural England, Defra and the Forestry Commission[39], Sen *et al.* find that both the number of trips and the value per trip to marine and coastal areas is higher than for most other types of land cover, including grasslands, mountains, or woodlands. The visitor number model is combined with a meta-analysis on the value per recreational trip across different types of habitats. The results show that the value per trip for marine and coastal areas is higher than for wetlands, freshwater and floodplain areas, woodlands and forest, but lower than for urban fringe farmlands. A follow-up analysis using these results was performed for coastal recreation in 26 counties along English coast (Schaafsma *et al.*, in prep). By multiplying the estimated number of visits by the value per trip to the coast (£4), the total annual value of visits to coastal sites was estimated at £39million. The original models by Sen *et al.* make no further distinction between different coastal margin habitat types, and are therefore less useful for informing scenario analyses that focus on specific coastal and marine habitats. Moreover, a number of assumptions had to be made to assign the original visited sites to various land use categories, and the value per trip is based on a meta-analysis of international recreation studies. The reliability of trip numbers and the value per trip may therefore be limited.

Three studies are available that assess values associated with **beach** recreation. Georgiou *et al.* (2000) use an open-ended CV survey to estimate public WTP for achieving compliance with the EC Bathing Water Directive to ensure safe bathing conditions at beaches in East Anglia. UK respondents, including residents, daytrippers and holiday makers, were interviewed at three locations (two seaside towns, Lowestoft and Great Yarmouth, and in Norwich) and asked about their WTP for benefits in terms of health risk reductions following the implementation of the Bathing Water Directive. Mean WTP was found to be approximately £49/hh. No distance-decay effects were assessed. The CV valuation scenario did not specify which risk reduction would be achieved, which makes it difficult to link the results to quality changes in scenario analysis. The study was discussed in a workshop (see Bann *et al.*, 2003) together with two other studies, and the workshop concluded that a bathing water study by eftec (2002) would be more suitable for national level assessments as the quality of the study was better and the sample representative for the English and Welsh population. However, the eftec study is not peer-reviewed and falls beyond the scope of this review.

Hanley *et al.* (2003) combine TC and Contingent Behaviour data to estimate the WTP for better coastal water quality at seven different beaches in Scotland. The results suggest a 1.3% increase in the number of trips should water quality improve to 'very good' standards, with an associated increase in consumer surplus of £7.66/pp or £0.63/trip. Using a population estimate of 661,110 people, this gives a figure of aggregate benefits of £1.65million/yr. This aggregate value estimate is uncertain, as it is not based on the total number of trips taken to beaches along the South-West coast, but on the average number of beach trips per person and the local population. It also does not reflect any current non-users that might visit beaches under improved conditions. Furthermore, the estimates are based on respondent-perceived quality (which drives visitation and values) rather than ecological quality indicators (which relate to policy objectives), and the values reflect the WTP for the change from the current quality to 'very good' quality both as perceived by respondents.

Finally, Bateman *et al.* (2001) look at the benefits of beach replenishment to avoid coastal erosion – and thereby obtain extra recreational possibilities in Caister-on-Sea, Norfolk. Using an open-ended CV survey, they find that holiday makers to the site are willing to pay £34/hh/yr, and local residents £41/hh/yr, based on non-parametric bootstrapping. These values result in an aggregate benefit of £971,640/yr. The authors argue that these benefits would outweigh the cost of beach replenishment.

[39] http://www.naturalengland.org.uk/ourwork/research/mene.aspx

The sample is relatively large (n=452) and the CV study designed according to state-of-the-art criteria. However, it is unclear on which population size (and characteristics) the aggregate value estimate was based.

Although these primary studies fulfil most standard reliability and validity criteria, the surveys were executed prior to 2000 and the use of these values in BT may produce less reliable results. A more recent, but rather specific CE study on **beach** amenities is presented in Christie and Gibbons (2011). The study assessed WTP for a change in coastal defences in Borth, North Wales, through repairing existing timber groynes, replacing them with rock groynes, or construct an offshore submerged reef, in combination with raising the existing seawall (potentially), leading to potential changes in beach safety and surfing conditions. The results show that WTP for safer beaches is £38/hh/yr, and the WTP for better surfing conditions is £16.5/hh/yr. The study was based on a sample of local residents (n=120), who were familiar with the beach and its conditions. The study is applicable to studies evaluating similar coastal defence systems, such as seawalls, rocky groynes or submerged reefs. However, the study does not give much detail on how the change in recreational amenities ('safer', 'improved wave quality for surfing') should be interpreted quantitatively so that they can be related to specific policy objectives, and it is therefore not clear what the change in ecosystem service provision is that is reflected in the WTP values.

One study falls into the **small islands** category. Chae *et al.* (2012) use TC to estimate the non-market recreational benefits arising from the Lundy Island Marine Nature Reserve in the UK. The estimated mean WTP for visiting Lundy was found to range from £420 to £672 per trip (per group of people). There are a number of probable explanations for this high value. Its protected status may make Lundy different from other small islands, which should also be taken into consideration when using the results for BT. The resulting demand curve, from which the consumer surplus values are derived, was very inelastic, i.e. the number of trips was relatively insensitive to price changes, which results in high value estimates. There were multipurpose trips where visitors had additional destinations besides Lundy, which were included in the value estimation. Further limitations of the study include the small sample, the exclusion of scuba divers from the sample, and the negative effect found for income which would suggest that tourism to Lundy is an inferior group, a finding that would not be expected from theory.

The use-values estimated in the CE presented in Luisetti *et al.* (2011) of **salt marshes** in the UK (see above) reflect a WTP value between £5,041 and £9,501/yr, depending on the size of the salt marsh (from 81.6ha up to 2,404.1ha) and the inclusion of non-use values. The CE attributes included the area of the new salt marsh, the distance from the respondent to the salt marsh, the presence of endangered birds, a dummy variable reflecting whether the marsh would be accessible to the public, and a tax increase. WTP for salt marsh re-creation is lower for people living further away from the salt marsh. WTP estimates also increase with the size of the marsh but in a highly non-linear way (by £1.36/hh/yr per ln(ha) increase). Respondents attribute higher welfare to salt marshes when they can recreate there (£4.91/hh/yr), as demonstrated by the significant 'access' attribute. Marginal WTP for the presence of observable protected birds is declining as the number of bird species increases, with £2.09/hh/yr for three additional species, and £4.06/hh/yr for five additional species. The sample included local residents of Norfolk, Essex and Suffolk, who were interviewed at train stations. The results of this study can be transferred to other sites if information about the population at the policy site is available. Given the attribute range in the CE, the study is most suitable for valuation of salt marsh creation with an extent between 81.6 and 2,404ha; for smaller and larger areas transfers become more complicated (see Luisetti *et al.,* 2013b). Similarly, given the attribute range in the CE, values can be applied to the population up to 72 miles from the new salt marsh.

Two studies assess marginal values for recreational activities at the **coastal shelf**. Bosetti and Pearce (2003) use a CV study about seal conservation in southwest England. The sample included 112 respondents interviewed at a Seal Sanctuary, and 94 respondents who went seal watching on an organised trip. The single bounded dichotomous choice WTP question resulted in a mean WTP of about £10 per person for the option of seeing seals in a specialised sanctuary for seals recovered from accidents, and closer to £12 per person for seeing seals in the wild. These values relate to recreational 'use' values, and the payment vehicle was an increase in entrance/trip prices. It is difficult to relate these values to a quantified change in seal population, although the authors mention that the seal population at the time of the study was estimated at 350-400 individuals, and this population was decreasing by 8-9% per year, whilst the Sanctuary saves 24-54 seal pups per year. Furthermore, the sample was relatively small.

The results of the CE about recreational **coastal** angling in southwest England presented in Lawrence (2005) show that WTP values per fishing trip varied by species. Respondents were asked to choose between two sites with different average catch rates of favourite and other species, size of fish, bag/rod limits, quality of the surrounding environment and the cost per day, along with the option to do something other than angling if none of the two options were suitable. For an 'all species' case, an increase by 50% in number of fish caught was valued at £7.65 and in fish size at £12.28; for bass the valuations were £10.15 (number) and £14.93 (size) respectively. The relationship between catch size and WTP is non-linear (declining). The results show that increasing the size of individual fish would have a larger impact on WTP than increasing the catch per day. WTP for 50% increase in number of cod caught is £7.61. For additional mackerel caught WTP equals zero because demand is satiated at the status quo level but for a 50% size increase WTP equals £11.14. For other species WTP equals £6.72, and size increases are valued at £14.93. All WTP estimates were expressed as a cost per day (i.e. per fishing trip). This study was based on a survey among 358 anglers at shore angling marks, angling competitions and in tackle shops in all parts of the south-west region, and may be biased towards summer anglers. The results presented Lawrence (2005) can be used in scenarios of change, as they reflect the values associated with specific changes in biophysical parameters.

Three studies assess the direct income earned in the **coastal margin** dependent tourism and recreation sector. Although these values indicate the economic importance of coastal recreation, the estimates are not directly related to changes in environmental quality or habitat extent and their use in scenario analysis would require additional assumptions. Moreover, they do not reflect consumer surplus, i.e. the welfare that people derive from coastal and marine tourism on top of what they have to pay on accommodation, transport, excursions, entrance fees, etc. In Parsons *et al.* (2003), the direct income revenues of whale-tourism in West-Scotland was estimated to be £2.3million/yr just from excursion tickets, and another £7.9million/yr is earned through tourist expenditures, including accommodation, food and travels. Results were based on interviews with boat operators, visitor-centre managers, tourists and local residents. There are a number of limitations to the results that may reduce the accuracy, including: boat operators did not provide financial estimates, so revenues were inferred from annual passenger numbers; there were no accurate figures for the total number of visitors at the survey sites, so the number of visitors to information centres was used instead; interviewed tourists 'frequently' misunderstood the survey question about their expenses on trips and accommodation.

Rees *et al.* (2010) estimate the expenditures on sub-aqua diving, sea angling and wildlife watching in Lyme Bay (southwest England) at £19.8million/yr. Of the sectors studied, sea anglers have the highest

total estimated expenditure per year of £4.8million/yr, compared with divers' expenditure of £1.1million/yr, and boat charter and dive businesses turnover of £3.8million/yr. The sample of interviewed anglers was relatively small (n=40) and based on a subsample of angling club members, while the majority of divers are non-members. The paper does not provide sufficient information to evaluate the assessment of divers' expenditure.

Ruiz Frau *et al.* (2013) assess the total revenues produced by diving, kayaking, wildlife viewing cruises and seabird watching in Wales. Divers, kayakers and seabird watchers were surveyed using an online questionnaire, whilst customers of wildlife viewing trips were surveyed by means of face-to-face questionnaires, which collected information on expenditure, demographics, and characteristics of the trip. The estimated total expenditure incurred by divers was around £8.4 million per annum. Using the average cost of a kayaking trip and the estimated number of activity days in Wales the annual expenditure associated to sea-kayaking in Wales was estimated at £2.7million (95% confidence interval (CI): £2.3M, £23.1M). The total expenditure incurred by boat passengers in Wales in 2008 on the day of the trip was estimated at £14.8million/yr (95% CI: £13.1M, £15.9M). As this expenditure was incurred on the day of the trip it can be considered that marine wildlife viewing was responsible for the majority of these costs. The total economic expenditure derived from seabird watching activity in Wales was estimated at approximately £3.7million/yr (95% CI: £3.1M, £4.8M).The value estimates should not be interpreted as the total economic value or coastal tourism in the study area, as the study excluded the assessment of surfing, sailing, yachting and shipping benefits, as well as recreational angling. Again, this assessment excludes consumer surplus.

Bateman *et al.* (2009) use a CE to estimate the WTP for land use changes, including the conversion of farmland to flooded mudflats. They find that the WTP for increased areas of **flooded mudflats** at high tide is negative, i.e. reflecting a welfare loss, even though ecologists consider these areas of high biodiversity habitat value. Unfortunately, the paper does not clearly present the values that should be used for transfer purposes.

Multiple studies are available for the UK published before 2000, which we list here. Penning-Roswell *et al.* (2006) present an overview of CV studies executed between 1988 and 1996 aiming to assess the WTP for coastal recreation. King (1995) presents a CV study on the benefits of beach recreation. Whitmarsh *et al.* (1999) present the results of a CV study on the value of enjoyment of a visit to the seafront at Lee-on-the-Solent near Portsmouth. Georgiou *et al.* (1998) assess the benefits of improved bathing water near Norfolk in a CV study. For the purpose of BT, we would, however, recommend using more recent studies, because the valuation literature has shown that value estimates are not stable over prolonged periods of time.

Aesthetic values as reflected in property prices

The only UK-based study has been developed for the UK NEA 2011. Mourato *et al.* (2010) find that house prices in England are not significantly associated with distance to the coastline or the availability of marine and coastal margins in the km^2 in which a house is located. However, it may be that the effect of seascape aesthetics on housing prices could not picked up at the coarse scale of this analysis and should not be considered conclusive evidence for the absence of aesthetic benefits reflected in GB housing prices. International studies (n=17) find evidence of the added value of nearby ecosystem services in house prices. Eleven of these studies concern beaches, of which seven are about sites in the USA and four in Spain. One study looks at the effect of tropical coral reefs on property prices in Hawaii (Cesar & van Beukering, 2004). For the other studies (related to estuaries, salt marshes and wetlands),

only USA studies were found. Given the large differences in housing markets between the UK and Spain and USA, transferring values to the UK is expected to generate large errors in value estimates.

Spiritual and cultural well-being and aesthetic benefits of wild species and seascapes

There are over 60 international valuation studies that address the economic welfare that people derive from biodiversity, species, habitat and/or landscape conservation. These reflect both spiritual and cultural wellbeing and aesthetic values. One CV study and four CE studies provide primary value estimates for the UK.

In their CV study on seal conservation in southwest England, Bosetti and Pearce (2003) also include a non-use component. Respondents (tourists) were asked to state choose their WTP amount from a payment ladder. The payment vehicle was an optional donation to a conservation organisation (in addition to current entrance fees) in order to mitigate conflicts between fishermen and seals and conserve seals in the wild. This value, not associated with actual viewing, was found to be £689,239, aggregated over half of the annual Seal Sanctuary visitors and based on the most conservative amount that respondents were almost certain they would pay (£3.55/pp on average). However, besides the relatively small sample, the payment vehicle employed for non-use values (donation) is not considered to be incentive compatible, i.e. not stimulating respondents to state their 'true' WTP, because they could avoid actual payments would the proposed donation request be implemented.

Luisetti *et al.* (2011) assess the WTP for **salt marsh** creation along the English coast using a CE. The obtained WTP for bird species (see above) is likely to reflect, at least in part, non-use values (see Luisetti *et al.,* 2013b). The study also shows that people are willing to pay for salt marsh creation even when they won't be allowed access to the site. However, reliable extraction of pure non-use values is not possible.

Birol and Cox (2007) use a CE to assess the WTP for otter hold creation and protected bird species. WTP values are found to be positive, estimated at £37.2/pp and £1.4/pp respectively, with an additional £13.8/pp/km^2 of **wetland** area. The latter value applies to wetland losses and gains between -147 and +100km^2, with a baseline of 247km^2. The sample contained both users and non-users, so the values for the ecosystem goods and benefits are not only non-use values. The values are based on a relatively simple conditional logit model, which current guidelines for CE would consider inferior to mixed logit models that account for panel effects and respondent heterogeneity. The sample is also small (n=100) which leads the authors to conclude that the results are indicative, but decision makers should use results based on larger samples to formulate socially efficient action plans.

McVittie and Moran (2010) use a CE to ask respondents for their WTP to install Marine Protected Areas (MPAs) in UK **(coastal waters).** The sample included coastal and non-coastal households in England, Wales, Northern Ireland and Scotland. Based on a mixed logit model, the estimated mean WTP to halt biodiversity losses is highest in Wales (£116/hh/yr) and lowest in Scotland (£23), compared with £37/hh/yr in Northern Ireland and £75/hh/yr in England. For increasing biodiversity mean WTP is highest in England (£75, showing no sensitivity to scope)[40] and lowest in Scotland (£26), whilst Welsh households are willing to pay £66/hh/yr and Northern Irish households £41/hh/yr. Part of these WTP values reflect use values. The levels of the attributes were defined as 'increase biodiversity' and 'halt loss of biodiversity', hence the change in ecosystem service provision is not described quantitatively (mainly because a lack of such information), which may limit the possibilities for BT.

[40] Sensitivity of WTP values to the scope (size or quality) of the good or benefit can be a relevant validity criterion for SP studies, where theoretically WTP is expected to increase with scope.

Ressurreicao *et al.* (2011; 2012) implemented a CV survey to assess the WTP for marine species among residents and visitors in three European **coastal** areas, including the Isles of Scilly. The WTP values were elicited using a payment card. The scenarios included a decline in the number of fish species, marine mammals, algae, sea birds and invertebrates (10% or 25%). The results show that the absolute WTP for the prevention of species loss, corrected for purchasing power, as well as the ranking of species, varies significantly between locations (the Azores in Portugal, Gdansk in Poland and the Isles of Scilly in the UK), but all values are around 2-3% of monthly household income. Scilly residents' WTP is lowest for invertebrates (£41), fish (£43) and birds (£44), and highest for marine mammals (£49) and algae (£53). The authors argue that this may be related to the prominence of seals and kelp forests around the Isles of Scilly. Visitors have lower WTP, with values of £36, £38, £39, £43 and £46 respectively. The study also provides confidence intervals for these mean WTP estimates, which reflect per household values as one-off contributions to a fund. The results did not show significant sensitivity to scope, i.e. losing fewer species was not associated with significantly higher WTP. The authors related this to warm glow effects in combination with limited understanding about the implications of species loss and ecological uncertainty about the effects of species loss on other communities. The protest bid rate in the Isles of Scilly (19%) is also higher than SP standards would normally allow.

There is one earlier paper on preservation. White *et al.* (1997) provide a study on the WTP for preservation of the otter and the water vole, which both inhabit coastal areas. However, these estimates are probably no longer reliable for BT purposes, as the original survey was conducted quite some time ago.

Education and research

The only available UK-study by UKMMAS (2010) reports that the Research Council's spending on marine science in 2007/08 was £75million. However, this is not a peer-reviewed study and does not present marginal values that can be associated with a change in the quantity or quality of coastal and marine habitats. Commenting that the value is likely an overestimate, Pugh and Skinner (2002), as reported in Beaumont *et al.* (2008), provide a figure for the UK of £317million (2002 values) reflecting marine research funding. Since the paper by Pugh and Skinner is not a scientific article, we exclude it from our analysis.

There are only two other, non-UK, academic studies that meet our study selection criteria and assess the economic value of education and research. While these studies are not directly applicable to the UK they do provide some notion of the magnitude of this category of benefit. Samonte Tan *et al.* (2007) estimate that the net annual revenue of education and research related to tropical coral reefs in the Philippines is about $32–$111/ha/yr), based on the costs of various education and research activities and facilities. Cesar and van Beukering (2004) assembled the annual budgets of research candidates involved in coral reef related research in Hawaii. The sum of these activities amounted to $10.5million in 2001.

4.6.4.4 Benefit transfer options

The gaps in the primary UK-based valuation literature limit the BT possibilities to inform management, especially for ecosystem goods and benefits and habitats that are considered to be important. Moreover, the available studies use different valuation methods and the results are not necessarily comparable. As discussed in the previous section, the studies also vary in terms of reliability and validity.

Table 4.14. shows that for some of the goods and benefits for which there are no primary UK studies available, there may be value estimates from other countries for BT purposes. However, value transfers across different countries can result in large errors, although these errors can be reduced by adjusting for income differences (Bateman *et al.*, 2011a). Such errors can be caused by differences in socio-demographic and economic characteristics of the population, as well as ecological and biophysical characteristics of the study sites (Brouwer, 2000). Ideally, transfer studies should therefore rely on multiple studies, either through meta-analytical function transfers (e.g. Brander *et al.*, 2012) or by providing a range transfer estimates using different primary studies (e.g. Luisetti *et al.*, 2011 for carbon). To what extent there are differences between different types of habitats and goods and benefits in transferability of economic value estimates has not been addressed (comprehensively) in the academic literature about coastal and marine ecosystem services.

Table 4.14. Overview of number of global studies for each combination of habitat and ecosystem service. The numbers refer to the number of studies that provide at least one value for the ecosystem service in a particular habitat type. Yellow indicates services for which one to four studies are available, and green indicates that five or more studies are available for a service in a habitat type.

	Pro-ducts	Sea defence	Erosion preven-tion	Healthy climate	Tourism and nature watching	Education research	Aesthetic: property	Spir./ aest.: wild species, seascapes
Dunes	0	4	0	1	3	0	1	1*
Beaches	0	3	1	0	43	0	11	9
Cliffs,small isl.	0	0	0	0	1	0	0	0
Machair	0	0	0	0	0	0	0	0
Lagoons	2	0	0	0	2	0	0	0
Salt marshes	2	6	0	4	5	0	1	4
Mudflats	0	2	0	2	1	0	0	1
Mangroves	12	5	2	0	3	0	0	2
Inter. wetland	2	1	0	0	5†	0	1	8
Seagrass beds	2‡	0	0	1	1	0	0	3
Kelp forest	3	0	0	0	0	0	0	0
Estuaries	5	0	0	1	12¶	0	2	5¶
Tr. Coral reefs	5	0	1	0	40	2	1	16
Cold coral reefs	0	0	0	0	0	0	0	0
Rocky bottom	1‡	0	0	0	1	0	0	0
Coastal shelf	16	0	0	2	30	0	0	24
Open ocean	0	0	0	1	0	0	0	1

* Nunes and Van den Bergh (2003) present a TC-CV study on the WTP to protect beaches in the Netherlands against algae blooms.
† Meyerhoff (2004) presents a CV study in Germany on the tourism benefits of the Wadden Sea.
‡ Stal *et al.* (2008) present a study on fisheries and the nursery function supporting commercial fisheries provided by seagrass beds and rocky bottom areas in Sweden.
¶ Atkins *et al.* (2007) (see also Atkins & Burdon, 2006) provide values for tourism (swimming, fishing, boating) and aesthetic/spiritual values of wild species in a fjord in Denmark.

As a rule of thumb, we suggest that for benefit transfer to the UK using international studies, studies from North- and West-Europe could be applied with the necessary caution, then studies from South- and East-Europe with more caution, followed by Australian and North-American studies with further increased caution, and studies from elsewhere should probably not be applied due to large differences in cultural, economic and ecological differences. There are four North- and West-Europe studies

published that provide values for habitat/good&benefit combination for which no UK studies are available, which we will mention here but not evaluate. Nunes and Van den Bergh (2003) present a TC-CV study on the WTP to protect beaches in the Netherlands against algae blooms. Meyerhoff (2004) presents a CV study in Germany on the tourism benefits of the Wadden Sea. Stål *et al.* (2008) present a study on fisheries and the nursery function supporting commercial fisheries provided by seagrass beds and rocky bottom areas in Sweden. Atkins *et al.* (2007) (see also Atkins & Burdon, 2006) provide values for tourism (swimming, fishing, boating) and aesthetic/spiritual values of wild species in a fjord in Denmark. These studies may provide an initial figure of the order of magnitude of values of the goods and benefits but are likely to arise in high errors given the differences in social and ecological characteristics and are probably insufficiently reliable for socially efficient and equitable decision making.

Transfer errors may also arise when studies are transferred over time. One of the fundamental assumptions in BT studies is that preferences underlying WTP estimates are robust over time (Brouwer, 2006). However, changes in respondents' socio-economic characteristics or other contextual factors, may alter preferences. When transferring value estimates or functions, underlying preferences are assumed to remain stable. In practice, study results have been transferred over long time periods to estimate the benefit values of ecosystem services at new policy sites. Empirical tests of temporal stability of SP studies for environmental goods and benefits based on CV studies indicate that choices are roughly consistent within short time periods (e.g. one year), but may change over longer periods of time. The same results have been found in health care studies when testing the transferability of CE results. We are not aware of any test-retest studies to test temporal stability of other valuation methods. Based on the temporal stability of SP work, one could consider the temporal cut off in our study selection (year 2000) somewhat weak. However, for practical reasons related to policy information provision, such a cut off is necessary to be able to provide at least some estimates of ecosystem service benefits.

4.6.5 Prioritisation of future research resources

The main objective of this literature review was to assess the extent to which existing valuation studies are available for the important coastal and marine habitats and the ecosystem goods and benefits they provide. For the different types of coastal margins, the UK NEA 2011 provides an assessment of the importance in terms of their contribution to human wellbeing of the various goods and benefits (or the amount of good/benefit delivery per unit area) that these habitats provide, as summarised in Table 4.11.3 of the UK NEA 2011 (Jones *et al.*, 2011). Table 4.11.3 was based on expert judgement, generated and agreed by the 14 authors of the UK NEA 2011 Chapter 11 who are all coastal experts, and went through a peer review process including 18 reviewers with different backgrounds, coastal and non-coastal. The UK NEA 2011 report does not provide a similar table to Table 4.11.3 for marine habitats. We first set out to develop an importance matrix for marine habitats (see **Appendix 4.7**), and we used this information to see if the existing literature covers the goods and benefits judged to be most important in the UK.

4.6.5.1 Valuation studies of important marine ecosystem goods and benefits

We combined information on the relevance of goods and services per habitat type (i.e. a more concise version of NEA 1 Table 4.11.3 (Jones *et al.*, 2010) and **Table 4.A4.** in **Appendix 4.7**) with the availability of UK valuation studies (**Table 4.12.**) to highlight important data gaps. This assessment is presented in

Table 4.15 for coastal habitats and **Table 4.16** for marine habitats. The number in each cell reflects the number of studies that are available for that particular good/benefit in the habitat. The colour coding shows the following:

- red: goods and benefits judged of high importance with no relevant valuation studies;
- orange: important goods and benefits with 1 valuation study, or goods and benefits of medium importance with no valuation studies;
- yellow: important goods and benefits with 2 or more valuation study or goods and benefits of medium importance with 1 valuation study; and
- white: goods and benefits of low importance or goods and benefits of medium importance with 2 or more valuation studies.

Table 4.15. Importance of ecosystem services per coastal habitat and the availability of UK-based valuation studies. Red: services of high importance with no relevant valuation studies; Orange: important services with one valuation study, or services of medium importance with no valuation studies; Yellow: important services with two or more valuation study or services of medium importance with one valuation study; White: services of low importance or services of medium importance with two or more valuation studies.

	Pro-ducts	Sea defence	Erosion preven-tion	Healthy climate	Tourism and nature watching	Education research	Aesthetic: property*	Spiritual/ aesthetic: wild species, seascapes
Dunes	0	2	0	1	1	0	0	1
Beaches	0	1	1	0	4	0	0	0
Sea cliffs	0	0	0	0	1	0	0	0
Machair	0	0	0	0	0	0	0	0
Lagoons	0	0	0	0	0	0	0	0
Salt marshes	1	4	0	4	1	0	0	1

* Property related aesthetic values are not included in Table 11.3 of UK NEA 2011.

Table 4.16. Importance of ecosystem services per marine habitat and the availability of UK-based valuation studies. Red: services of high importance with no relevant valuation studies; Orange: important services with one valuation study, or services of medium importance with no valuation studies; Yellow: important services with two or more valuation study or services of medium importance with one valuation study; White: services of low importance or services of medium importance with two or more valuation studies.

	Products	Sea defence	Erosion preven-tion	Healthy climate	Tourism and nature watching	Education research	Aesthetic: property	Spiritual / aesthetic: wild species, seascapes
Mudflats	0	2	0	2	1	0	0	0
Inter. wetland	0	0	0	0	0	0	0	1
Seagrass beds	0	0	0	0	0	0	0	0
Kelp forest	0	0	0	0	0	0	0	0
Estuaries	0	0	0	0	0	0	0	0
Cold water coral reefs	0	0	0	0	0	0	0	0
Rocky bottom	0	0	0	0	0	0	0	0
Coastal shelf	3	0	0	1	5	0	0	5
Open ocean	0	0	0	0	0	0	0	0

As the many red and orange cells in **Table 4.15.** indicate, there are considerable gaps in the UK valuation literature related to ecosystem goods and benefits provided by coastal margins. Sea defence benefits have received very little recent attention despite the long UK coast line and the increasing risk of flooding due to sea level rise. Other than the study by Bateman *et al.* (2001) that assesses the benefits of prevention of coastal erosion by quantifying the recreational values protected by a shingle beach, the economic value of erosion prevention provided by coastal habitats in the UK has not been addressed in any academic publication since 2000. Climate change related benefits of carbon sequestration have only been assessed for dunes and salt marshes, whereas other coastal habitats are judged to be at least of medium importance.

Moreover, provisioning services related to land-based activities on coastal margins, including the production of crops, meat, wild food, wool, reed, grasses, timber and turf, require more attention (see

Table 11.3 in Jones *et al.*, 2010). The studies in our global valuation dataset also do not provide value estimates for these goods and benefits from coastal habitats.

There are no studies for the (flow of) goods and services provided by machair even though this is a unique type of habitat and only found in the UK and Ireland, and considered to be very important for sea defence, recreation, education, cultural wellbeing, aesthetics and biodiversity. For saline lagoons, there are no studies either, whilst they contribute to human wellbeing especially in terms of tourism and recreation. Cold water coral reefs have not been addressed in the UK yet, although one study has been conducted but not published in the peer-reviewed academic literature (Jobstvogt *et al.*, 2013).

In **Table 4.16.**, we cross-tabulate the availability of valuation estimates and the importance of ecosystem goods and benefits for the marine habitats, using the same colour coding. Similar to the findings for coastal margins, there are many 'important' ecosystem goods and benefits for which no or few estimates are available (cells coded red and orange). This holds for education and research related values, but, perhaps more importantly, for spiritual and cultural wellbeing and aesthetic values of seascapes (including those reflected in property values), and products and other raw materials related benefits including the nursery function that supports this provision. To fill the gaps in the literature and be able to project ecosystem goods and benefits flows in scenario analyses, future studies should also provide more insight into sustainable harvesting levels, analyse the value of fisheries net of other capital inputs, and include the economic value of other raw materials, including seaweed and pharmaceuticals.

Tourism values are currently being assessed within the UK NEAFO for a range of substrate/ habitats, including rocky seafloors with shell beds, large kelp, seaweeds and sea-pens, and sandy and muddy sea floors with different types of plant growth, including soft corals and sponges (see Kenter *et al.*, 2013). The combined CE-CV study provides positive WTP estimates for MPA development, which appear to depend on habitat. Preliminary results also show positive effects for sites where seals, octopus and birds may be encountered.

Table 4.16. suggests that more valuation efforts should be directed towards intertidal wetlands and estuaries. Contrary to the international valuation literature, there are no UK primary valuation studies that focus on estuaries (but see Turner *et al.*, 2007 for a benefits transfer-based analysis for the Humber estuary), whilst their provision of products and different cultural services (tourism, education and research, aesthetic values of species and seascapes) are considered to be important in terms of their contribution to human wellbeing. However, for estuaries and other 'habitat complexes' or 'habitat mosaics', it may be possible to use valuation studies for other types of habitats that are present in the estuary (or habitat mosaic) of interest. Habitats found in estuaries include mud flats, salt marshes, rocky shores and beaches. The same may hold for intertidal wetlands, because this type of habitat includes other types, such as mudflats, mangroves and salt marshes. Not all goods and benefits of mudflats and salt marshes are well covered and their value in terms of biodiversity conservation in particular deserves more attention. Moreover, the biophysical ecosystem service provision level as well as the economic values for the associated benefits may not be independent from the adjacent habitats within a habitat mosaic. In the presence of synergistic or antagonistic effects of one habitat type, fragmented within the mosaic, on the delivery of any particular service from another interspersed habitat type may not have the same value as a single block of habitat of equivalent overall size.

The nature of valuation studies to fill the gaps depends on the policy-issue at stake (i.e. how the value estimates will be used): its scale (local, regional, national), the required level of accuracy, the dominant ecosystem services and associated value types (use vs non-use values) as well as the available budget for

primary data collection. For example, for national assessments such as the UK NEA broadly generalisable values are appropriate, for example based on large scale household surveys, whilst for more regional policy issues such as managed realignment where local sensitivity is important to consider, a more targeted assessment among local key stakeholders may be more useful. Monetary valuation mainly supports economic analysis, such as CBA. Where cultural assets and spiritual values are relevant, social impact analysis through wellbeing and 'shared value' assessments may require deliberative approaches and citizen/stakeholder forums (see **Section 4.7**), sometimes in combination with monetary valuation exercises.

4.6.5.2 Availability of value estimates and policy needs

There is a multitude of policies governing coastal and marine habitats (see Boyes & Elliot, in prep). This includes international obligations that the UK has committed itself to, such as the OSPAR convention, the UNFCCC and Kyoto Protocol on climate change, waste related policies such as the London convention and Ballast water convention, the Convention on Biological Diversity and the Convention on International Trade in Endangered Species for biodiversity, etc. These international obligations have been translated into EU Directives, including the WFD and MSFD, the CFP, Floods Directive, Habitat and Birds Directives, and Strategic Environmental Assessments and Environmental Assessment regulations. Many of these policies are linked or overlap.

The UK Marine Policy Statement highlights that marine plans need to take, among others, the WFD and MSFD into account to achieve sustainable development. The MSFD aims to achieve GES of the EU's marine waters by 2020. The definition of this status can vary by region. The MSFD does not prescribe any measures, except for the establishment of MPAs. Measures must be based upon an impact assessment including a cost-benefit analysis. As such, economic valuation plays a key role. The WFD and MSFD together cover all coastal and marine habitats and therefore economic value estimates are required for all types of habitats. The UK Biodiversity Action Plan (UKBAP) has defined five priority habitats: coastal sand dunes, coastal salt marsh, coastal vegetated shingle beaches, maritime cliff and slopes and machair. Coastal and saline lagoons are included under the UKBAP priority marine habitats, together with 24 others habitats. It is therefore difficult to prioritise research efforts based on policy needs based on habitats or ecosystem goods and benefits.

4.6.6 Concluding remarks

Clear gaps have been identified in this review exercise for UK coastal/marine ecosystem valuation data. A number of important habitats, ecosystem services and related goods and benefits have few or no valuation estimates assigned to them (e.g. estuaries), while other politically sensitive goods and benefits such as sea defence and erosion prevention have a database dominated by results from studies conducted a decade or more ago. While benefits transfer may offer some pragmatic assistance to cover a limited number of the gaps, this procedure is unlikely to be any sort of panacea. Both temporal and cultural bias constraints remain formidable challenges for any benefits transfer exercise using data more than a decade old and spatially more distant than a rough boundary around Northern Europe. The available literature provides only four North-European studies which may only fill a few gaps. The only real exceptions to this rule are global benefits such as those related to carbon sequestration and storage. It might also be argued that non-use existence values for values associated with wild species diversity may have limited transferability but the concept itself and its reliable valuation via SP studies is still very much an open research question.

The obvious conclusion form this review analysis is that more primary valuation research needs to be undertaken. **Tables 4.15.** and 4.**16.** offer some guidance on the foci for this possible new research programme. Highlighted gaps include the sea defence and coastal erosion prevention benefits, as well as climate benefits and provisioning services (products) provided by coastal habitats. Goods and services provided by saline lagoons, machair and cold water corals have not been assessed at all in the published academic literature. For marine ecosystem services, more valuation studies may be required for aesthetic values and spiritual and cultural wellbeing from seascapes and wild species diversity, as well as products and other raw materials, education and research. Finally, the complexity of 'mosaic' habitats, such as intertidal wetlands and estuaries, may require valuation studies that consider these in aggregate terms, rather than trying to disentangle the values goods and benefits provided by subhabitat types independently and at the same time avoding double counting.

Monetary valuation of coastal/marine ecosystem goods and benefits are a tool in the DSS, and aims to support the step of economic (and social) appraisal and valuation of changes in ecosystem services flow and different management options in adaptive coastal management. In absence of monetary values, comparing different options and trading off gains of some ecosystem goods and benefits against the losses of others becomes more complicated without valuation evidence. The economic appraisal will be limited or partial, and the outcome of CBAs may be biased, but consideration of 'unvalued' goods and benefits can take place either qualitatively or within a further social assessment in a balance sheet approach, as described in **Section 4.7**.

4.7 Appraisal Format: the 'balance sheet' approach

Any DSS that is put in place to assist in evaluating the gains and losses involved in marine planning and management will need to encompass a wide diversity of impacts and different stakeholder perspectives. The balance sheets approach set out below (see **Figure 4.45.**) is a pragmatic attempt to provide a framework within which the complexity of real world decision making and trade-offs can be examined and presented. It sets out three complementary components (balance sheets) which can be seen as 'roughly comparable' sets of findings with overlaps and linkages. The aim would therefore be to determine the 'best' combination of data, methods and analysis, depending on the actual activity and context under appraisal (Turner, 2011).

The complexities and the non-commensurate values that characterise the real world political economy of 'contested' natural resource allocation and trade-offs are clearly illustrated in fisheries policy. The annual fisheries negotiations in Brussels try to set rules for fair access to fish stocks. Scientists have recommended total limits to catch to avoid fishing beyond levels that the stock will support. Ministers then meet together at the annual Fisheries Council to set pragmatic rules of access based on instruments such as gear type, number of vessels, days at sea and total allowable catch. In the past, ministers have often negotiated catch allocations that exceed the advice of their own scientists. One reason for this is the non-commensurability of the currencies used by different sectors engaged in the process, each of which seeks to archive 'sustainability'. The fleet owners seek to sustain profits (market values, that can be subjected to a CBA); local political representatives seek to optimise or conserve employment and multiplier effects at the community level (measured as jobs and susceptible to financial impact analysis at a local scale); and conservationists emphasise non-use values and ethical considerations (more amenable to deliberative methods including MCA). The Minister at the Council tries to balance these interests but, without an effective analytical framework, and with competing claims from other ministers, the likelihood of success is quite low. The next reform of the EU CFP will try to improve this situation by following the 'Ecosystem Approach' that recognises humans as an intrinsic part of the system and that total allowable catch or damage to habitats and non-target species cannot be permitted to exceed ecosystem limits.

Another policy context concerned with coastal protection and sea defence also highlights the 'wicked' characteristics common in many environmental management situations. Over the past decade or so UK government policy in terms of future investments in coastal management has been re-orientated away from a 'hold the line' philosophy and towards a more flexible approach. The new approach has included coastal realignment schemes in selected locations and also a greater recognition of coastal processes such as erosion and subsequent beach replenishment. But the DSSs and policy planning had not been sufficiently adjusted before the headline strategic policy shift became widely publicised and stakeholder concerns were raised. Poor policy support sequencing has meant that difficult 'local' policy impacts and controversies have been raised and policymakers have been slow to respond. Thus the switch towards a more flexible coastal management regime can be justified on overall cost grounds and national strategic requirements, together with a precautionary approach to possible climate related sea level rise and storm intensity and frequency predictions. But the distributional consequences should have been recognised in advance of the policy switch, and mitigation measures should have been in place, as well as a more targeted information and awareness campaign. Instead the agencies involved have had to play catch up, following numerous stakeholder protests and campaigning and wide press coverage. So acceptable 'compensation' measures for the 'losers' in any given coastal scheme (and for that matter flooding risk situations more generally across catchments) have only slowly emerged as controversy has

escalated. The pathfinder scheme trialled in East Anglia, for example, has examined a number of compensation measures for householders affected by coastal erosion. Under a 'balance sheets' DSS the distributional impacts and 'local' impacts would have been diagnosed prior to the strategic policy switch and arguably more effective ameliorative measures would have been in place.

In the balance sheets approach, three types of complementary assessments (balance sheets) are envisaged to try to give some guidelines for steering a reasonably objective course through these 'contested' policy contexts, and these are illustrated in **Figure 4.45.**:

1. economic (monetary) CBA using a conventional economic efficiency criterion (macro UK economy efficiency), but augmented with a distributional analysis of impacts and possible equity weighting;
2. regional and local financial impacts and policy analysis, covering impacts like local unemployment, loss of community identity and related financial multiplier effects which often raise issues of compensation; and
3. trade-off analysis (non-monetary) better suited to dealing with collective or shared values across wider society such as, for example, intrinsic value in biodiversity, cultural assets value etc.

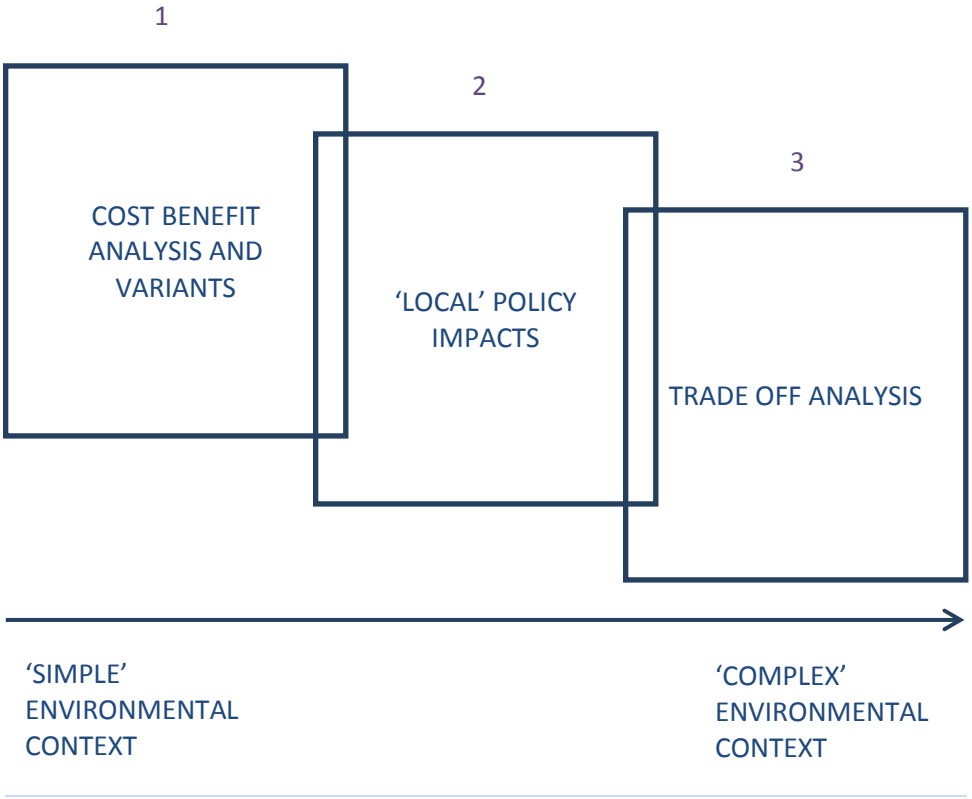


Figure 4.45. Balance sheets appraisal method.

The analytical sequence of the balance sheet approach would begin with an economic cost-benefit scoping analysis and then proceed to include the other balance sheets depending on the context and the type of decision under scrutiny. The aim would not be to aggregate the results of each balance sheet but to present the policy process with the set of linked findings in as transparent a way as possible.

Given the range of data that relates to the marine environment and related socio-economic activities, there is a pressing need to agree broad categories of data which can illuminate the economic, social and environmental dimensions of environmental change in the marine context. The balance sheets approach aims to achieve this by separating out, in the first instance, economic data and analysis. So in the first column of **Figure 4.46.**, economic data is covered and is guided by the criterion of macro-economic efficiency and informed by market-based data, WTP data and cost data (including second best data such as GVA etc). A key link to the second column in **Figure 4.46.** is provided at the bottom of the first column when the issue of the distribution of costs and benefits is raised i.e. who gains and who losses from any change.This forces the analyist to think through feasible and necessary compensation measures. In MSFD terms we are now including certain elements of social appraisal in economic appraisal (enabled via CBA or CEA). The second column of **Figure 4.46.** now expands on the sort of data and issues that are best classified as social effects (including equity and fairness impacts) with a spatial boundary (local/regional) condition imposed on the analysis. The final column of **Figure 4.46.** continues the social analysis but now encompasses values and impacts that are often expressed at the national scale and are contested with a variety of underlying ethical criteria. Clearly the columns overlap but the aim is to give some logical sequence to a decision support method(s)/ process which is trying to scope and analyse real world economic and socio-political issues on a rough scale from relatively 'simple' to very complex 'wicked'situations.

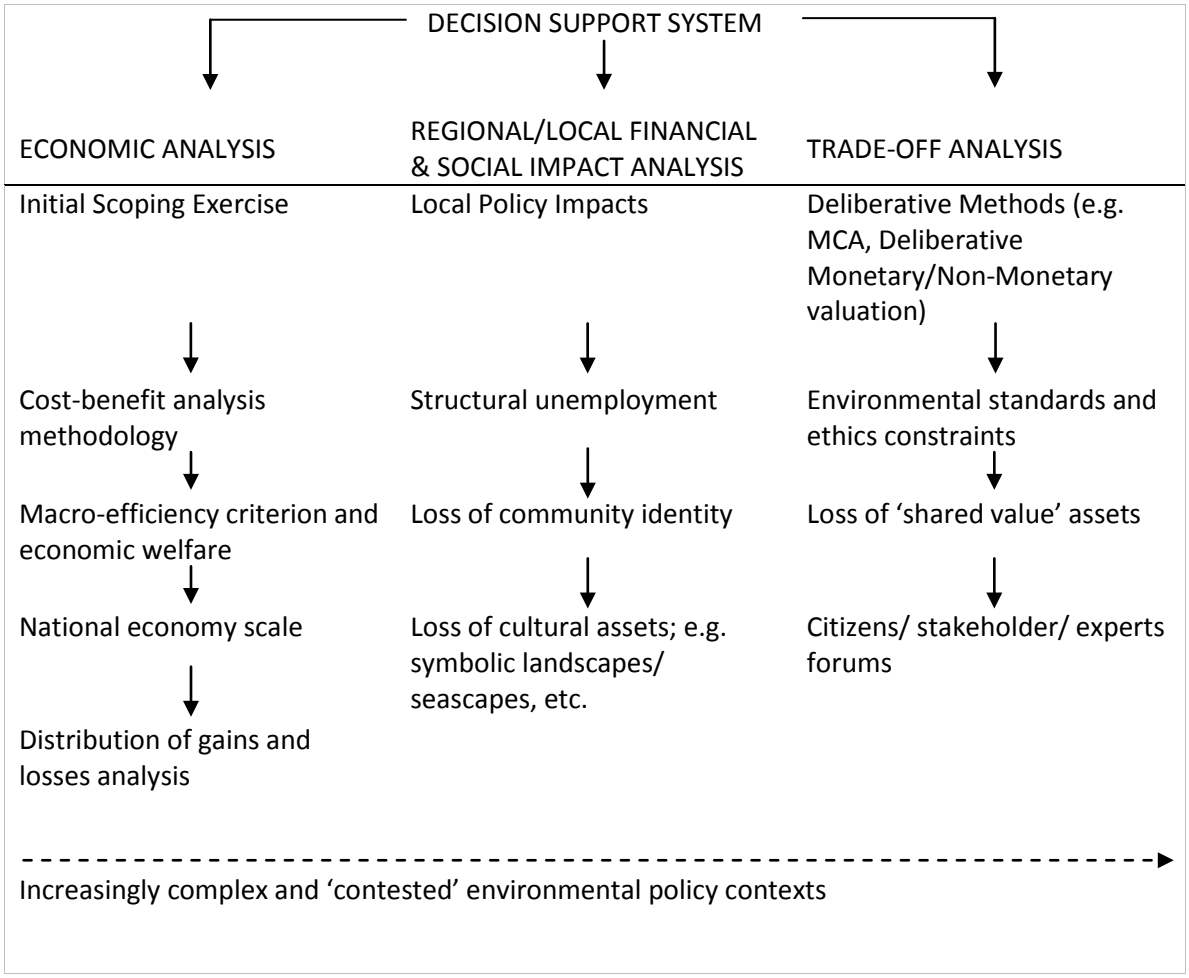


Figure 4.46. Policy Appraisal Balance Sheets.

In ideal circumstances, the framework of action to deliver sustainable management needs to fulfil a set of tenets covering all facets of decision making and the identification of defendable sustainable development measures, especially in 'wicked' policy contexts (Elliott, 2011, and references therein). These indicate that our actions are required to be environmentally or ecologically sustainable, economically viable, technologically feasible, socially desirable or tolerable, administratively achievable, legally permissible and politically expedient. These seven tenets (Elliott, 2011) have been augmented by a further three tenets: ethically defensible (morally desirable), culturally inclusive and effectively communicable (Elliott, 2013). This is a formidable list of requirements and pragmatism rather than a futile search for meta-ethical perfection is the recommended course of action under AM. But following this guidance almost inevitable means trade-off choices and therefore winners and losers. The exact combination of decision criteria and support tools that are relevant will depend on the prevailing and expected policy context and the type of trade-off. The heavy, extensive and on-going utilisation of coastal and marine resources ensures that management decisions will be contested by competing interests. The goal of a return to good (pristine) conditions (Hering *et al.*, 2010) is also unlikely to prove practicable, and so the DSS and social dialogue has to focus on the future and feasible future environmental system states.

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4.10 Appendices

Appendix 3.10.1 Discounting and ethics

We first focus on fairness across time (intergenerational equity) and the practice of discounting. The standard CBA practice of positive, fixed and short term (<25years) discounting does not sit easily within policy contexts with pressures and drivers such as climate change and related global economic forces. A growing number of analysts agree that discounting at a constant and relatively high (i.e. determined by reference to market interest rates data) rate of discount over time horizons of 100 years or more is problematic. The effect is to make even large costs or benefits incurred in the distant future seem inconsequential and this feels intuitively wrong (Weitzman, 1998). There seems to be a tyranny imposed by current generations on the future when, for example, there is current inaction conditioned by cost considerations and a neglect of low weighted future benefits, e.g. climate change (Groom *et al.*, 2005). Asheim (2012) has recently summarised the dilemma in the following way: contemporary society needs to distinguish between what the current generations as a collective should do ethically to serve the interests of all generations from an impartial perspective, and what contemporary countries or individuals should do strategically to serve their own interests when such actions influence the strategic action of other countries and individuals.

The ethical and strategic dilemma is not as straightforward as it may appear upon superficial examination. Zero or negative discounting poses a threat to the least well off in today's society and can result in large sacrifices from the present for the benefit of later generations who may be better off. A single invariant low rate of discount could in some circumstances allow a greater volume of projects to pass the CBA test and therefore strain resource or environmental capacities. Nevertheless, some modifications to the standard CBA discounting procedure have been adopted in UK public sector project appraisal (HMT, 2003). A time declining discount rate (DDR) procedure over at least 100 years is now recommended for projects with significant environmental impacts.

A range of reasons have been put forward in support of DDR, including uncertainty about future interest rates and the macro-economic state of the economy (Weitzman, 1998; Gollier, 2002). Some empirical evidence exists for 'hyperbolic discounting' indicating that individuals value medium and distant futures on an equivalent basis, i.e. the discount rate falls the longer the time horizon. It may be that individuals live in relative not absolute time and therefore revise and re-evaluate plans continually as time passes; or over time individuals pass through different stages of life and change as people (Henderson & Bateman, 1995; Heal, 1998; Frederick *et al.*, 2002). Advocates of the conventional discounted utilitarian approach in conventional CBA would, however, counter that social discounting as practiced by governments should not mimic the 'time inconsistent' or 'irrational' behaviour of individuals exhibiting hyperbolic discounting behaviour. But while policy inconsistency at a given period of time is an institutional failure and should be corrected, policy switching over longer periods of time are inevitable and necessary if future uncertainties and 'surprises' are unavoidable. Finally, Knetsch (2005) has claimed that individuals discount future losses at a lower rate than the value of future gains and that therefore rates reflecting observed individual preferences would give more weight to future environmental losses, justify greater current sacrifices to deal with them and support policies that reduce the risk of future loss.

The ethics versus strategic behaviour dichotomy is the focus for a key set of arguments. If individual people also have (individual/shared) other-regarding (social) preferences and if they trade off their own

material interests against the wider interests of society, then some notion of fairness that captures social preferences is required. Roemer (2011) has argued that the utilitarian social welfare function used in conventional CBA assumes that the decision problem for a society with many generations is equivalent to the decision problem of an infinitely-lived consumer. This claim is refuted by Dasgupta (2011), but both agree that the discount rate based on market data and applied to the climate change problem is too large. They disagree on what is the better ethically defensible sustainability criterion, with Roemer favouring the Rawlsian intergenerational maximin approach, i.e. each generation passes on a non-declining, in per capita welfare terms, capital (human, physical and natural capital components) bequest. Roemer argues that consumption as conventionally defined in economics is not the only component of welfare or wellbeing. Educated leisure, quality of the local to global environment and knowledge are also direct inputs into welfare. Intergenerational maximin is not problem free (e.g. how much sacrifice is a fair burden for the current generation rich and poor?), but in the spirit of moral pluralism (i.e. there is no meta-ethical criterion and the context and consequences of ethical choices should not be ignored) intergenerational maximin may still prove to be a usable ethical guide.

Finally, Asheim (2012) has proposed what he calls an equity-rank-discounted utilitarian intergenerational equity position in which welfare is discounted not according to time but according to rank. This approach it is claimed can combine equal treatment of generations with social discounting by giving priority for the worse off not only due to their absolute level of wellbeing (Rawlsian-maximin) but also their relative rank in wellbeing. If the future is better off than the present, then this criterion is on a par with discounted utilitarianism. However, if, for example, climate change brings an end to the past positive correlation between time and increasing welfare, then rank-discounted utilitarianism makes a greater call for present action (and lower discount rates) to protect the interests of future generations. In the marine environment scientific work has shown that key processes are slow with timescales over 1000 years or more, for example, with ice sheet or deep ocean changes. So changes in current policy related to economic development and climate change could have impacts stretching out 1000 years (Stouffer, 2012). But will this make the future worse off than contemporary society? Taking a precautionary approach, the 2007 IPCC (Intergovernmental Panel on Climate Change) sea level rise predictions (maximum 2 foot rise in sea level by the end of the century) now seem too optimistic as they failed to factor in ice sheet melting impacts. Now some estimates put the sea level rise up to 7 feet (Young and Pilkey 2010), and some coastal authorities have design plans with 2.5 feet (in the Netherlands) and 4.6 feet (in California, USA) rise parameters built into them to correct for the low IPCC 2007 estimates (*ibid.*). While ice sheet melting is non-linear and difficult to predict, the threat posed to human welfare is significant. Low elevation coastal zones i.e. contiguous areas with elevations below 10m, contain 10% of the global population and have expanding populations, and a large proportion of the world's megacities.

Appendix 3.10.2 Total Economic Value

Total Economic Value (TEV) decomposes into use and non-use (or passive use) values but it does not encompass other kinds of values, such as intrinsic values which are usually defined as values residing 'in' the asset and unrelated to human preferences or even human observation. However, apart from the problems of making the notion of intrinsic value operational, it can be argued that some people's willingness to pay (WTP) for the conservation of an asset, independently of any use they make of it, is influenced by their own judgements about intrinsic value. This may show up especially in notions of 'rights to existence' but also as a form of altruism.

Coastal and marine ecosystems provide a wide range of final services and related benefits of significant value to society - fisheries, transport medium, storm and pollution buffering functions, flood alleviation, recreation and aesthetic services. The use of the TEV classification enables the values to be usefully broken down into the categories shown in **Figure 4.A1.** The initial distinction is between individual *use value* and *non-use value*. Use value involves some interaction with the resource, either directly or indirectly:

- d*irect use value*: involves direct interaction with the ecosystem itself rather than via the services it provides. It may be consumptive use, such as fisheries, or it may be non-consumptive, as with some recreational and educational activities. There is also the possibility of deriving value from 'distant use' through media such as television or magazines, although it is unclear whether or not this type of value is actually a use value, and to what extent it can be attributed to the ecosystem involved;
- i*ndirect use value*: derives from services provided by the ecosystem. This might, for example, include the removal of nutrients, thereby improving water quality, or the carbon sequestration services provided by the ocean or some coastal ecosystems;
- non-use value is associated with benefits derived simply from the knowledge that a particular ecosystem is maintained. By definition, it is not associated with any use of the resource or tangible benefit derived from it, although users of a resource might also attribute non-use value to it. Non-use value is closely linked to ethical concerns, often being linked to altruistic preferences, although according to some analysts it stems ultimately from self-interest. It can be split into three basic components, although these may overlap depending upon exact definitions;
- e*xistence value*: derived simply from the satisfaction of knowing that an ecosystem continues to exist, whether or not this might also benefit others. This value notion has been interpreted in a number of ways and seems to straddle the instrumental and intrinsic value divide;
- b*equest value*: associated with the knowledge that a resource will be passed on to descendants to maintain the opportunity for them to enjoy it in the future; and
- a*ltruistic value*: associated with the satisfaction from ensuring resources are available to contemporaries of the current generation.

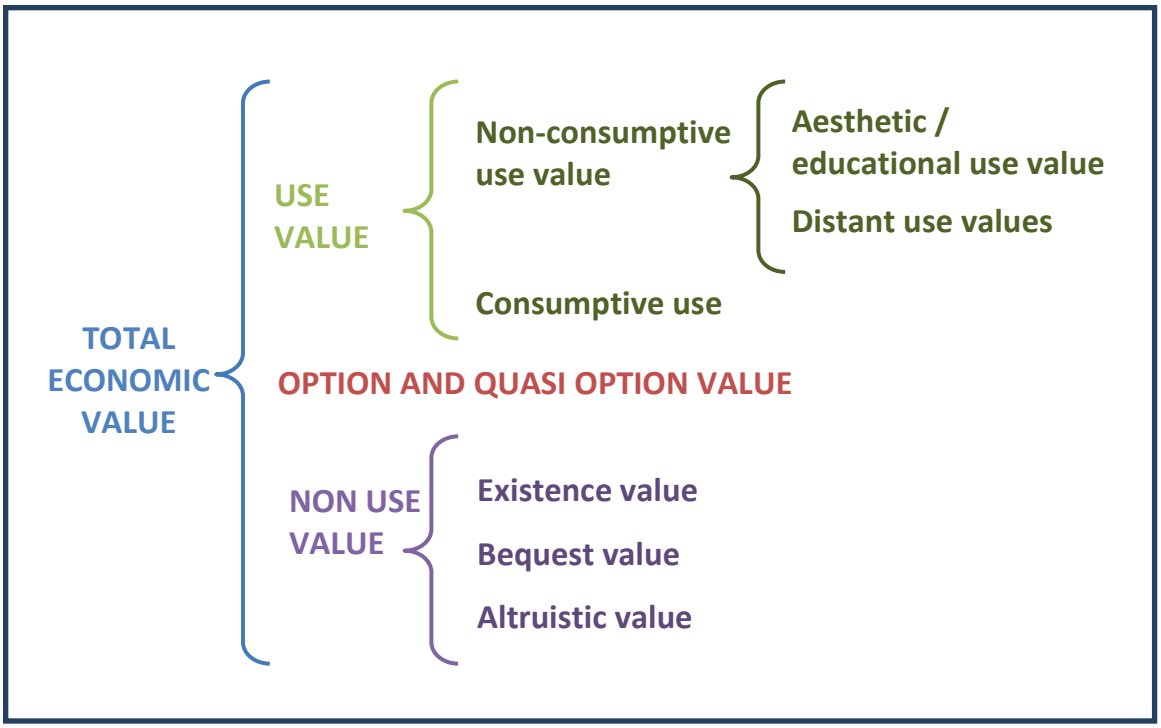

Figure 4.A1. Total Economic Value.

Finally, two categories not associated with the initial distinction between use values and non-use values include:

- o*ption value*: an individual derives benefit from ensuring that a resource will be available for *use in the future*. In this sense it is a form of use value, although it can be regarded as a form of insurance to provide for possible future but not current use; and
- q*uasi-option value (QOV):* associated with the potential benefits of waiting for improved information before giving up the option to preserve a resource for future use. In particular, it suggests a value of avoiding irreversible damage that might prove to have been unwarranted in the light of further information. An example of an option value is in bio-prospecting, where biodiversity may be maintained on the off-chance that it might in the future be the source of important new medicinal drugs. Potentially, QOV could make up a sizeable proportion of TEV, although measurement of its magnitude is problematic.

These various elements of total economic value are assessed using economic valuation methods, and some of these elements are more easily valued than others, especially those with easily identifiable uses (usually the use type values). Non-use values are usually more difficult to assess.

Financial versus economic values

In any socio-economic assessment it is necessary to distinguish between financial accounting and economic values and analysis. Prices and values are not necessarily equivalent and price is only that portion of the underlying value of a good which is realised in the market place (Pearce & Turner, 1990). For those goods produced and consumed under reasonably competitive market conditions, their prices are an acceptable approximation for their value, provided that there are no other prevailing distortions such as government tax or subsidy interventions. Prices will typically diverge from values when so-called public goods (non-exclusion and non-rivalry in consumption characteristics) are involved which lack private ownership; or when the full costs of production and consumption (especially environmental

impact costs) are not readily included in the pricing process. For many ecosystem (service-related) goods there are no markets available, or the full cost of their supply are not reflected in financial measures. Economic analysis seeks to uncover the value in monetary terms (and ultimately the economic welfare effect on humans) of the good in question rather than just its financial price. It measures value (welfare) through an approximation known as WTP for changes in the provision of the good. Note that this WTP measure is not the same thing as actual payment (market price); when the latter is less than the former a consumer gains value (consumer surplus).

Appendix 3.10.3 Indicators of coastal/marine ecosystem services and the MSFD

The MSFD has stimulated work into targets and indicators linked to the 11 descriptors of marine ecosystem change which encompass the cause and consequence of human activities on ecosystem service provision (Borja *et al.*, 2010; Cochrane *et al.*, 2010; Defra, 2012; ABPmer & eftec, 2012; Borja *et al.*, 2013). Over 70 indicators have been identified at the EU-level for guiding progress to achieve Good Environmental Status (GEnS) (**Table 4.A1. –** European Commission, 2010) although the means of combining these indicators into an integrated assessment of GES have still to be established (Borja *et al.*, 2013). It is argued that the MSFD requires functional indicators, for example, across media, spatial location, hydrological function and biological function. As a means of determining the GEnS of the marine environment, it is suggested that the 11 descriptors are hierarchical and relate to the functioning of the system and that some, such as biodiversity and food-web structure, integrate across the other descriptors (Borja *et al.*, 2010) (**Figure 4.A2.**). The ecosystem service indicators presented here (**Tables 4.4.-4.7.**) are broadly consistent with those GEnS indicators given in **Table 4.A1.**

Table 4.A1. MSFD descriptors and EU-level indicators (European Commission, 2010).

Descriptor	Criteria	EU Indicator
1. Biological diversity	1.1. Species distribution	1.1.1. Distributional range
		1.1.2. Distributional pattern within the latter, where appropriate
		1.1.3. Area covered by the species (for sessile/benthic species)
	1.2. Population size	1.2.1. Population abundance and/or biomass, as appropriate
	1.3. Population condition	1.3.1. Population demographic characteristics (e.g. body size or age class structure, sex ratio, fecundity rates, survival/ mortality rates)
		1.3.2. Population genetic structure, where appropriate
	1.4. Habitat distribution	1.4.1. Distributional range
		1.4.2. Distributional pattern
	1.5. Habitat extent	1.5.1. Habitat area
		1.5.2. Habitat volume, where relevant
		1.5.3. Condition of the typical species and communities
	1.6. Habitat condition	1.6.1. Relative abundance and/or biomass, as appropriate
		1.6.2. Physical, hydrological and chemical conditions
	1.7. Ecosystem structure	1.6.3. Composition and relative proportions of ecosystem components (habitats and species)
2. Non-indigenous species	2.1. Abundance and state of non-indigenous species, in particular invasive species	2.1.1 Trends in abundance, temporal occurrence and spatial distribution in the wild of non-indigenous species
	2.2. Environmental impact of invasive non-indigenous species	2.2.1. Ratio between invasive non-indigenous species and native species
		2.2.2. Impacts of non-indigenous species at the level of species, habitats and ecosystems
3. Fisheries	3.1. Level of pressure of the fishing activity	3.1.1. Fishing mortality (F), as compared to Fmsy
		3.1.2. Catch/biomass ratio (where F is not available)
	3.2. Reproductive capacity of the stock	3.2.1. Spawning Stock Biomass (SSB), as compared to SSBmsy
		3.2.2. Biomass indices
	3.3. Population age and size distribution	3.3.1. Proportion of fish larger than the mean size of first sexual maturation
		3.3.2. Mean maximum length across all species found in research vessel surveys
		3.3.3. 95 % percentile of the fish length distribution observed in research vessel surveys
		3.3.4. Size at first sexual maturation, which may reflect the extent of undesirable genetic effects of exploitation
4. Food webs	4.1. Productivity (production per unit biomass) of key species or trophic groups	4.1.1 Performance of key predator species using their production per unit biomass (productivity)

Descriptor	Criteria	EU Indicator
	4.2. Proportion of selected species at the top of food webs	4.2.1. Large fish (by weight)
	4.3. Abundance/ distribution of key trophic groups/species	4.3.1. Abundance trends of functionally important selected groups/species
5. Eutrophica-tion	5.1. Nutrients levels	5.1.1. Nutrient concentration in the water column
		5.1.2. Nutrient ratios (silica, nitrogen and phosphorus), where appropriate
	5.2. Direct effects of nutrient enrichment	5.2.1. Chlorophyll concentration in the water column
		5.2.2. Water transparency related to increase in suspended algae, where relevant
		5.2.3. Abundance of opportunistic macroalgae
		5.2.4. Species shift in floristic composition such as diatom to flagellate ratio, benthic to pelagic shifts, as well as bloom events of nuisance/toxic algal blooms (e.g. cyanobacteria) caused by human activities
	5.3. Indirect effects of nutrient enrichment	5.3.1. Abundance of perennial seaweeds and seagrasses (e.g. fucoids, eelgrass and Neptune grass) adversely impacted by decrease in water transparency
		5.3.2. Dissolved oxygen, i.e. changes due to increased organic matter decomposition and size of the area concerned
6. Sea floor integrity	6.1 Physical damage, having regard to substrate characteristics	6.1.1. Type, abundance, biomass and areal extent of relevant biogenic substrate
		6.1.2. Extent of the seabed significantly affected by human activities for the different substrate types
	6.2. Condition of benthic community	6.2.1. Presence of particularly sensitive and/or tolerant species
		6.2.2. Multi-metric indexes assessing benthic community condition and functionality, such as species diversity and richness, proportion of opportunistic to sensitive species
		6.2.3. Proportion of biomass or number of individuals in the macrobenthos above some specified length/size
		6.2.4. Parameters describing the characteristics (shape, slope and intercept) of the size spectrum of the benthic community
7. Hydrography	7.1. Spatial characterisation of permanent alterations	7.1.1. Extent of area affected by permanent alterations
	7.2. Impact of permanent hydrographical changes	7.2.1. Spatial extent of habitats affected by the permanent alteration
		7.2.2. Changes in habitats, in particular the functions provided (e.g. spawning, breeding and feeding areas and migration routes of fish, birds and mammals), due to altered hydrographical conditions
8. Contami-nants	8.1. Concentration of contaminants	8.1.1. Concentration of contaminants in the relevant matrix (such as biota, sediment and water)

Descriptor	Criteria	EU Indicator
	8.2. Effects of contaminants	8.2.1. Levels of pollution effects on the ecosystem components concerned, having regard to the selected biological processes and taxonomic groups where a cause/effect relationship has been established and needs to be monitored
		8.2.2. Occurrence, origin (where possible), extent of significant acute pollution events (e.g. slicks from oil and oil products) and their impact on biota physically affected by this pollution
9. Contaminants in food	9.1. Levels, number and frequency of contaminants	9.1.1. Actual levels of contaminants that have been detected and number of contaminants which have exceeded maximum regulatory levels
		9.1.2. Frequency of regulatory levels being exceeded
10. Marine litter	10.1. Characteristics of litter in the marine and coastal environment	10.1.1. Trends in the amount of litter washed ashore and/or deposited on coastlines, including analysis of its composition, spatial distribution and, where possible, source
		10.1.2. Trends in the amount of litter in the water column (including floating at the surface) and deposited on the sea- floor, including analysis of its composition, spatial distribution and, where possible, source
		10.1.3. Trends in the amount, distribution and, where possible, composition of micro-particles (in particular micro- plastics)
	10.2. Impacts of litter on marine life	10.2.1. Trends in the amount and composition of litter ingested by marine animals
11. Energy (noise)	11.1. Distribution in time and place of loud, low and mid frequency impulsive sounds	11.1.1. Proportion of days and their distribution within a calendar year over areas of a determined surface, as well as their spatial distribution, in which anthropogenic sound sources exceed levels that are likely to entail significant impact
	11.2. Continuous low frequency sound	11.2.1. Trends in the ambient noise level within the 1/3 octave bands 63 and 125 Hz (centre frequency) measured by observation stations and/or with the use of models

The MSFD indicators presented in **Table 4.A1.** were not directly identified in relation to marine ecosystem services and goods/benefits. However, the relationship between the MSFD indicators and ecosystem service provision is under investigation by Cefas who are updating an earlier investigation by ABPmer and eftec (2012). This earlier investigation is based on a baseline BAU scenario of the marine environment, which implies that no additional actions or management measures under the MSFD are implemented, and identifies the links between MSFD GEnS descriptors and indicators and ecosystem services. The report also provides information on how a change in the state of the environment (identified through the GEnS descriptors) influences a change in the state of ecosystem services.

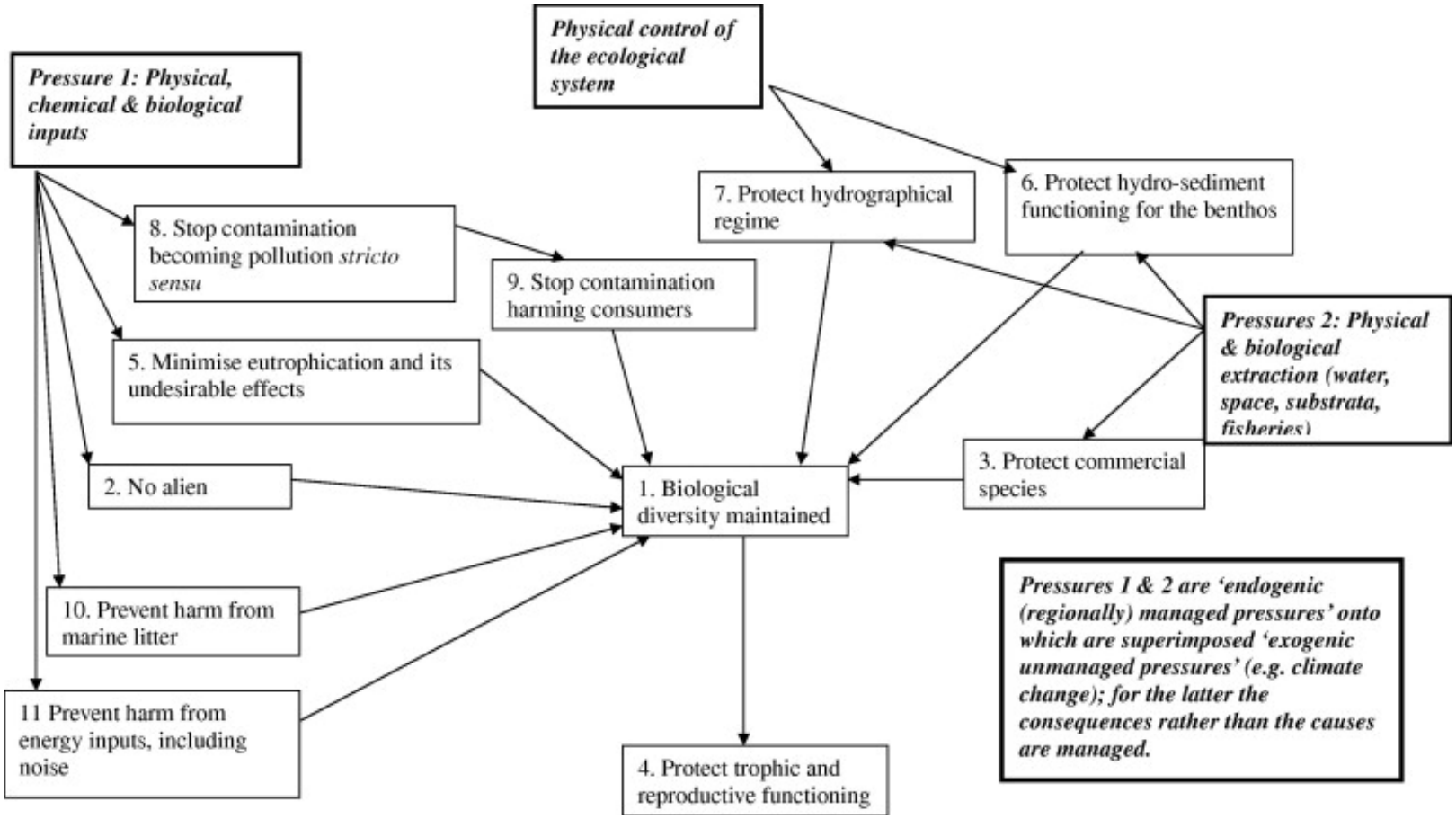

Figure 4.A2. A conceptual model showing how the MSFD interlinks pressures and the 11 qualitative descriptors. Source: Borja *et al.* (2010). Reproduced with permission of Elsevier.

Appendix 3.10.4 Model reliability, examples and terminology

Box 4.A1. Reliability and cost-efficiency of models

Models for estimating the impact of changing human pressures on marine ecosystem services need to be both reliable and cost-effective. They also need to be trusted. These attributes may be thought of as, respectively, the scientific, economic and social components of model engineering, which also requires the formulation of good algorithms for equation solving, and their accurate implementation in model code. This box will not further discuss the engineering issues, but will consider in turn reliability, resource cost, and trustworthiness. **Box 4.A2.** explains some modelling terminology and approaches.

A reliable model is one that, when properly initialized and parameterized, is able to simulate ecosystem states that match, within acceptable limits, the corresponding states of the target system.[42] When information about such states is already available, the simulations are *hindcasts*; prediction or forecast deals with future states, and the expectation is that when data become available about these states, a simulation made by a reliable model will be found to be in good agreement. There is a difficulty, however, when we turn to the simulation of scenarios. A model might, for example, forecast disastrous rises in sea-level if present trends in the emission of GHGs are allowed to continue; if these forecasts help in changing the trends, it will (fortunately) never be possible to assess whether the simulations were reliable. Nevertheless, the principle of assessing reliability by comparing simulation and observations remains true (the model would be proven reliable had emissions been sustained and sea-level risen), but there is an overlap with trustworthiness: does society trust the model (and the modellers and the modelling process) sufficiently to suffer the inconvenience of taking steps to avoid such disastrous sea level rise?
As **table 4.A1** shows, there are numerous dimensions to model reliability.

As **Table 4.A2.** shows, there are numerous dimensions to model reliability. The most important dimensions are (i) those relating to complexity, and (ii) the distinction between empirical and mechanistic models. Complexity increases with number of processes represented, number of state variables simulated, and numbers of parameters to be given values. Because ecosystems are complicated, it might be thought that adding state variables, and additional process parameterizations, to models, would improve their reliability. But this is not necessarily the case. Uncertainty in parameters can combine so as to degrade the performance of complex models below that of simpler models (e.g. Fulton *et al.,* 2003). The Akaike Information Criterion (AIC: Akaike 1973) offers a way to assess performance by correcting estimates of model goodness of fit (to a set of observations) for the number of parameters adjusted to get that fit. The criterion is perhaps a modern version of Occam's razor, which cautions us not to multiply entities unnecessarily.

[42] A typical requirement is that the difference between the simulation and the observations shall be much smaller than the overall variation in the observations. This practical requirement should be distinguished from the epistemological requirement (for a model to be considered valid) that chance or alternative models are unlikely to explain an agreement between simulation and observations.

Table 4.A2. Issues affecting model reliability and cost-efficiency

Issue	Options	Discussion
Model purpose and modelling strategy	Models as 'miniatures of nature' or models as tools for specific purposes	A detailed and reliable model of the North Sea ecosystem, for example, could be used to examine how any and all services respond to change; but such general purpose models don't exist as reliable predictive devices; instead models are built and used as tools, to explore specific issues (e.g. how to prevent eutrophication).
Type of processes described by model	Physical, biogeochemical, ecological, socio-economic	Physical models more reliable than biogeochemical models, and so on, because of lower levels of system complexity and greater importance of conservation laws (Tett & Wilson, 2000); application to ecosystem services typically need either dynamic (feedback) or pipeline coupling between these levels of models
Applicability and use of conservation laws	Conservation of energy and mass (in general), of chemical elements (e.g. C, N, P), of information and probability, of the value of a currency	Where such laws exist, models should be formulated to make use of them, thus constraining behaviour and increasing reliability; compare, for example, a N-limited algal bloom model (with maximum algal biomass constrained by total N) with a Lotka-Volterra predator-prey population model constrained only by choice of parameter values
Idealisation	Aristotelian (selection of processes deemed relevant) and Galilean (simplification of equations used) (Frigg & Hartman, 2006)	Making such choices is both the 'art of science' in forming refutable hypotheses and the 'art of model engineering' in identifying key process to include for a given purpose, and in 'parameterising' these processes once identified.
Number of model state variables and parameters	Few to many	Complex systems typically need more state variables to fully capture their state, and to simulate it with a model, and each process description requires one or more parameters with values to be specified; because of uncertainties in parameter values and in initial conditions for state variables, a model with 2*N* state variables is not necessarily twice as good as one with *N* state variables. The Akaike Information Criterion (AIC: Akaike, 1973) provides a means to identify optimum complexity. See also Stephens *et al.* (2006).

Issue	Options	Discussion
Spatial extent of modelled domain, and boundary conditions	From arbitrary boundaries to boundaries designed to minimise need for boundary condition data	A model domain will typically embrace the target system, such as the North Sea, but wise boundary placement increases reliability for a given amount of model complexity: boundaries are best sited where there are low or constant fluxes, or where there are data, or simply far enough away from the target area to make sure boundary effects do not impact much on the target system, or within which the system can be treated as homogenous; where boundary conditions cannot be supplied from observation, the model may need to be linked into a model with a more extensive or global domain (e.g. physical models for UK shelf seas linked to models for the N. Atlantic circulation).
Graininess and spatial dimensionality	0D (point or box models), grids in 1, 2 or 3D	Gridded models can more realistically simulate heterogeneous spatial conditions but require sophisticated algorithms to describe exchange between grid points or elements without incurring numerical errors; the chosen grid must be consistent with the scale of the phenomenon studied, as described by the equations, requiring parameterisation of sub-grid-scale processes (e.g. turbulence as eddy viscosity and diffusivity); the grid set up needs to represent fine scale topography for a fine scale application, and computation may be costly as number of spatial dimensions and grid-points increases.
Equilibrium versus dynamical models	Equilibrium models (independent of time while forcing constant); worst-case or best-case solutions (independent of time while forcing constant); dynamical models (involving one or more equations in dX/dt or $\partial X/\partial t$)	Equilibrium or static worst-case models often quick and easy to apply, but there might be more than one equilibrium solution, or the dynamics of change might be more important than the final state; dynamical models may need more data on initial conditions, sophisticated algorithms for numerical integration of (partial) differential equations and appropriate parameterisation of processes.
How to simulate systems with feedback loops	Linear or branching-chain cause-&-effect models; system models	Except for sets of equations with chaotic properties, models that link processes into a linear sequence (e.g. $A \rightarrow B \rightarrow C$) are easier to parameterise, validate and use than models with feedback loops (C is connected back to A) although such feedback can be very important; complex systems of this sort are sometimes said to have 'emergent' properties that may be difficult to capture in some numerical models.

Issue	Options	Discussion
Epistemological basis	Mechanistic models; empirical models	Mechanistic models embody scientific understanding of the underlying processes (obtained by experiment and hypothesis refutation over a range of conditions), and should be more reliable for extrapolation beyond the range of conditions under which they have been tested; explicitly empirical models (e.g. linear regression), or science-based models in which many parameters have been 'tuned' to a particular data set, should be extrapolated with caution.
Reducing uncertainty, increasing confidence	Ensemble models (multiple simulations with range of parameter values); multi-model ensembles	Dynamical mechanistic models for the same process may involve different decisions about idealisations and different choices for dealing with grid-point exchange and sub-grid-scale parameterisation, and so their simulations may diverge even when starting from a common set of individual values; climate modelling recognises this by presenting an envelope of results from multi-model ensembles (Collins, 2007; Meehl *et al.*, 2007)
Validation of models (and further estimation of limits of confidence in prediction)	Compare simulation with a set of observations independent of those used to test modelled hypotheses, to 'tune' model parameters, or to fit empirical model to data (if no independent data available, use a 'boot-strapping' method)	So long as their numerical algorithms have been correctly devised and programmed, mechanistic models can be considered to be *a priori* valid (because of their epistemological basis); nevertheless, added confidence can be given by comparing simulations with observations made outside the original domains used for research and model parameterisation ; the comparison of model hindcasts with time-series of historic data has proven convincing in climate modelling (e.g. Stott *et al.*, 2000); in the case of empirical models, which have been parameterised by fitting, testing against independent data sets is vital

The cost of a model is not only that of its engineering (devising its algorithms and writing its code) but also that of obtaining parameter values and boundary conditions. For example, a simulation with a 3D coupled physical-ecological model of the North Sea requires information about sea-bed depth and time-series of meteorological values at each model grid-point, initial conditions and ocean boundary grid-point time-series or no-flux assumptions for each state variable, and relevant land-discharge fluxes including freshwater and nutrients. The resolution and quality of these inputs influences the cost/effort involved in running the model and the quality of the output, all of which needs to be fit for and appropriate to the purpose intended. **Table 4.A3.** below gives some idea of the costs of model development and model application for some of the models discussed here, and which span a range of model types and complexities.

Finally, the question of stakeholder trust in models and their results, has been little explored. Under 'science-push', models were developed as part of scientific research, for example into marine ecosystems, and then offered to society as a solution to problems of resource usage. Two generations ago, the public were in general willing to trust 'the experts', but this is no longer true. A generation ago the 'science-push' model gave way to 'policy-pull' with the imposition of the 'customer-contractor principle', but this assumed that the customer knew what they wanted and could adequately specify a required model at the start of a contract (Boden *et al.*, 2006). The European project SPICOSA explored a third way to build models, involving continuing collaboration with stakeholders and policy makers (**Box 4.3.**). Although 'further research is necessary', the project's findings suggest that involving stakeholders and policy-makers with experts in the devising of conceptual models of a particular 'issue', leads to greater trust in the resulting models, especially when these are *models of intermediate complexity* that can be quickly assembled from 'off-the-shelf' scientific knowledge and algorithms.

Table 4.A3. Order-of-magnitude costs of developing and using some example models. Does not include any costs of underpinning scientific research)

Model (type)	Number of state variables and (typical) spatial compartments	Person-years for development	Example of an application	User time for this application
CSTT (simple, mechanistic)	2 and 1	about 3	assessing impact of wastewater discharge on estuary (Painting *et al.*, 2007)	month
ACExR-LESV (intermediate complexity, mechanistic)	8 and 3	about 10	assessing nutrient-assimilative capacity of a fjord (Tett *et al.*, 2011a)	month
ERSEM + physical transport (complex, mechanistic)	~ 60 (ERSEM ~ 55 currently, hydrodynamical model ~ 5)	10	simulating nutrient transport in the North Sea (Lenhart *et al.,* 2010)	months
Ecopath (complex,mixed mechanistic and empirical)	10 and 1	10	estimating potential fish take in a regional sea	months
typical SPICOSA SAF model (intermediate complexity, mixed)	1-10; a few	1-3 years*	optimising clam harvest in a lagoon (Canu *et al.*, 2011)	2-3 years*
global coupled ocean-atmosphere model (complex, mechanistic)	10-30; up to 10^6	10-100	simulating climate change in the UK (Jenkins *et al.*, 2009)	A few years

* In the SPICOSA case a model is developed from 'off-the-shelf' components for the desired application: the application time is the same as the development time

Box 4.A2. Two simple mechanistic models.

These models illustrate some of the issues relating to model development, reliability and cost, and provide a context for definitions of the terms *state variable*, *parameter* and *boundary condition*.

A. Model for the surface temperature of a planet with an atmosphere containing greenhouse gases
This model, which can be tested by comparing its estimates of mean planetary surface temperature with presently observed values on Earth, Mars, etc, can be used to understand and predict temperature change. It is a *mechanistic model* because it has been developed from well-validated physical laws (http://en.wikipedia.org/wiki/Idealized_greenhouse_model). It is an *equilibrium model* because its key underpinning assumption is that the planet is in thermal balance, i.e. on average, the inflow of short-wave solar energy is the same as the outflow of long-wave radiation.

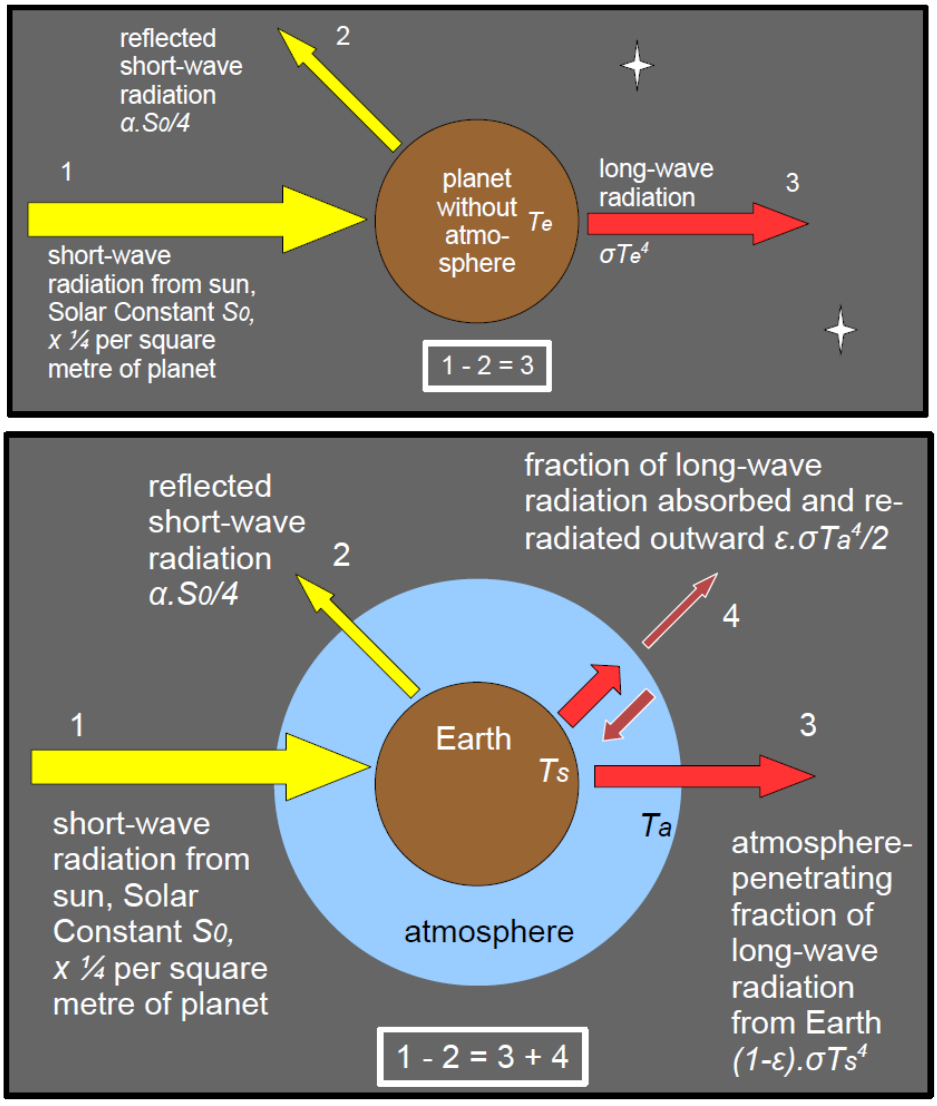

Figure 4.A3. Illustrating the simple model for planetary surface temperature, above without an atmosphere, below with an atmosphere.

Atmospheric GHGs interfere with the outflow, requiring surface temperatures to be higher to achieve the thermal balance than on a similar planet without an atmosphere. The outgoing radiant flux is assumed to obey the Stefan-Boltzman law, i.e. to depend on the 4th power of absolute temperature. The incoming flux is calculated from the solar constant (the intensity of sunshine observed outside the atmosphere) and simple geometric considerations (the surface area of a sphere is 4 times its cross-section). The equation for planetary surface temperature is:

$$T_s = T_e \cdot \sqrt[4]{\left(1 \Big/ \left(1 - \varepsilon \Big/ 2\right)\right)} \qquad \text{where} \quad T_e = \sqrt[4]{\frac{S_0\left(1-\alpha\right)}{4\sigma}} \text{ °K (without an atmosphere).}$$

In this equation there are two *state variables.* T_s is the actual planetary surface temperature, and T_e is the temperature that the surface would have if there were no atmosphere. These two values completely specify the state of the system described by the model.

The *boundary condition* is the solar radiation above the atmosphere, given by S_0. Although named the solar *constant*, with a present annual mean value of 1366 Wm^{-2} for Earth, it varies over millennia because of changes in planetary orbit. The model *parameters* are the other terms in the equations:

- σ = Stefan-Boltzman constant, 5.670373 × 10^{-8} kg s^{-3} K^{-4};
- α = planetary albedo, about 0.3 in the case of Earth, may change with ice and cloud cover;
- ε = the fraction of outgoing long-wave radiation from the planetary surface that is absorbed by atmospheric GHG, about 0.78 for Earth before widespread use of fossil fuels.

Of these, the Stefan-Boltzman constant is a universal, whereas the other two depend on conditions on the planet and in its atmosphere. Identifying them as *parameters* means they are given a fixed value for a particular solution of the model equations. The solution for our own planet, before extensive industrialization and use of fossil fuels, was 288°K, or 15°C.

Why does this model not suffice to predict the effect of anthropogenic GHGs? One important reason is that it does not take account of feedback loops. Planetary warming might decrease albedo because of ice-melt, or increase it through increased cloud cover caused by increased evaporation from the oceans. The second process might also multiply the warming effect of extra CO_2 because water vapour itself is a greenhouse gas. Thus more elaborate models have been developed that include descriptions of these processes, and which further take account of the circulation of heat and water around the planet: these are the GCMs, or *Global Coupled Ocean-Atmosphere General Circulation Models (*e.g. Stott *et al.*, 2000), which require advanced programming skills, powerful computers, and many expert judgements about values of multiple parameters.

They are consequently expensive to build and operate compared to simple models, but valuable particularly compared to the costs of observational programmes and remediation action , and are capable of supplying forecasts on the regional scale (e.g. the UK HADCM3 model suite: Murphy *et al.,* 2009).

B. The Comprehensive Studies Task Team (CSTT) screening model for eutrophication

This model was developed in response to the requirement of the Urban Waste Water Treatment Directive that 'comprehensive studies' be carried out on water bodies receiving a waste discharge and at risk of eutrophication (CSTT, 1994; Tett *et al.*, 2003). The water body, perhaps an estuary, is treated as a single well-mixed box, exchanging with adjacent waters.

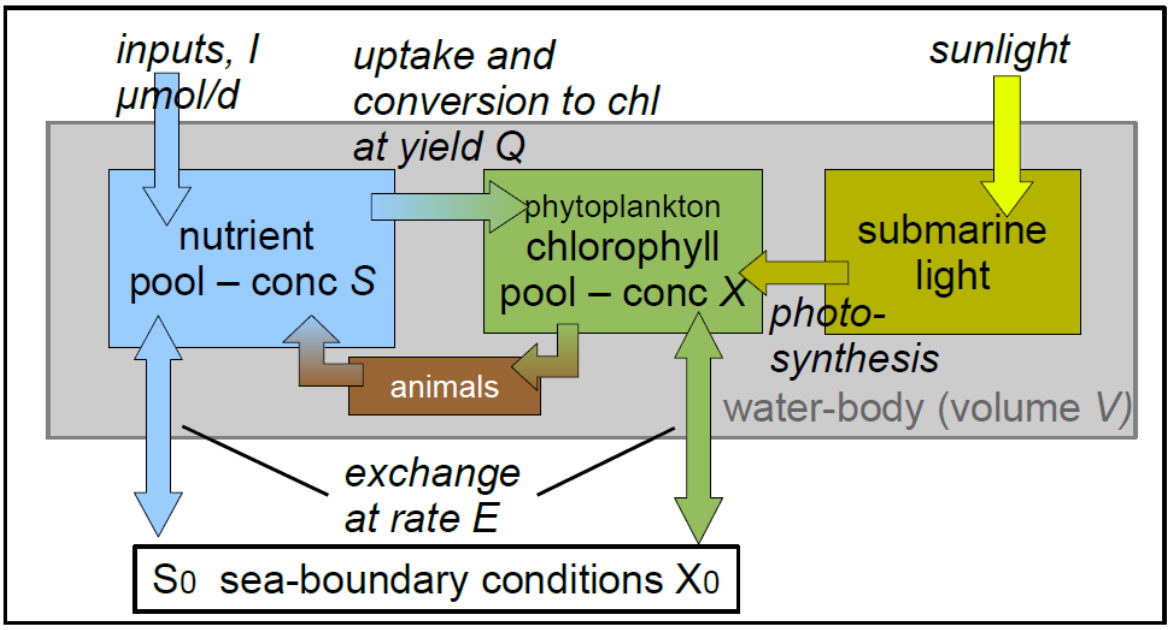

Figure 4.A4. A simplified conceptual model of an aquatic ecosystem, showing it as a single box exchanging with boundary conditions that are unaffected by the exchange.

• The diagram shows a very simple ecosystem model within the box. The CSTT model in essence answers two questions: what is the equilibrium nutrient concentration inside the box? answer:

$$S = S_0 + \frac{I}{E \cdot V}$$ µM

• what is the greatest concentration of chlorophyll that could occur, if all available nutrient were converted into chlorophyll, without loss to animals and without limitation by lack of light? answer:

$$X_m = X_o + Q \cdot S$$ µg/L

This is a *worst-case* model because calculated X_m can be compared with a legally-defined threshold value; if less, then there will never be a risk of eutrophication (i.e. of excess growth of algae etc. leading to an undesirable disturbance). The model has two *state variables*, neither of which can always be observed: nutrient concentration *S* (in the absence of phytoplankton), and maximum chlorophyll concentration X_m (requiring many observations before the true maximum is seen). The model's *boundary conditions* are the concentrations of nutrients and chlorophyll in the external sea (in the case of an estuary), and the nutrient input flux *I* µmol d^{-1} including the waste discharge of interest.

The model *parameters* are:
• *E* = exchange rate, d-1, the probability that a given water packet (containing nutrient and phytoplankton) will be exchanged with a sea-water packet during 24 hours;
• *V* = volume of the water body, in Litres (for consistency with the concentration units);
• *Q* = yield of phytoplankton chlorophyll from nutrient, µg μmol^{-1}.
Exchange rate and volume are properties of water-bodies, whereas the yield is somewhat more universal - i.e. a single value can be used for many water-bodies. It was originally estimated empirically (Gowen *et al.*, 1992) and then confirmed by experiments in mesocosms (Edwards *et al.*, 2003). However, both *E* and *Q* are extreme Galilean idealisations, the simplest imaginable descriptions (or

parameterisations) of more complex processes, including tidal exchange and estuarine circulation in the case of the physical property *E*, and algal nutrient uptake and chlorophyll synthesis in the case of the biogeochemical and algal physiological property *Q*. Finally, the CSTT model as a whole is a severe Aristotelian idealisation, excluding all but a few key ecosystem processes.

When these idealisations are considered to be too far from reality, or when it is required to simulate dynamical changes during a seasonal cycle, more elaborate models have been constructed. The ACExR-LESV model for simulating the impact of aquaculture in fjords, describes a physical system of three layers responding to forcing by tides, freshwater and wind stirring (Gillibrand *et al.*, 2013), LESV and a biological system that includes several kinds of algae described by a dynamic version of the CSTT model equations including control of growth by light as well as nutrients (Portilla *et al.,* 2009). ACExR- might be called a *model of intermediate complexity*.

ERSEM, the European Regional Seas Ecosystem Model (Baretta, 1997; Baretta *et al.*, 1997) describes a much more complex biological system. Initially it was implemented in a small set of boxes representing different parts of the North Sea, with boxes divided into near-surface and near-bed waters. Exchange between the boxes was provided from simulations with a hydrodynamic model programmed on a 3D spatial grid. More recently, ERSEM, or its derivative BFM, the Biogeochemical Flux Model, has been directly coupled to hydrodynamic models in 1D (Blackford *et al.*, 2004; Vichi *et al.*, 2004) and 3D (van Leeuwen *et al.*, 2012). As with GCMs, models like ERSEM, and the physical models used to drive them, are more expensive (than simple models) to construct and operate than simpler models, but potentially they provide outputs that will more closely describe reality.

Appendix 3.10.5 Water quality regulation – services and benefits

Coastal and marine habitats provide supporting services related to nutrient retention and waste assimilation that contribute to water quality control. In the ecosystem assessment framework used in the UK NEA 2011, which focuses on goods and benefits, such quality improvements are expressed as increases in the recreation, amenity and biodiversity values, as well as fisheries values and any other goods and benefits that depend on water quality. The importance of marine water quality regulation for drinking water is very low in the UK due to the excessively high costs of desalinisation. The availability of water for shipping etc. is regarded as an abiotic component of the marine system.

Economic valuation approaches become relevant with respect to water quality regulation objectives. When there are policies that have been adopted by the government and that focus on a particular water quality level, the main policy issue is to decide on the most cost-efficient way of achieving that water quality level. The implicit assumption is that by accepting the quality objective, society has expressed their preference for achieving the objectives and its associated welfare implications (a bundle of goods and benefits that depend on water quality levels).

A global meta-analysis on wetlands (Brander *et al.*, 2006) did not find a significant effect of water supply or water quality related goods and benefits. Mangi *et al.* (2011) use an avoided cost approach to value the water purification benefits of the **coastal zone** of the Isles of Scilly (UK), leading to an estimated £259,365/yr. This estimate is based on the assumption that the coastal waters provide similar treatments services as an artificial treatment plant without further negative environmental impacts, and a cost of treating 1kg BOD of sewage of £2.39 from South West Water. However, the costs of BOD treatment may vary widely between companies. Beaumont *et al.* (2008) estimate the value of nutrient cycling using the one-off costs of treatment of the entire volume of territorial waters of the UK, but their replacement cost estimate is based on Costanza *et al.* (1997) and is therefore excluded from our analysis.

Appendix 3.10.6 Carbon burial and sequestration – biophysical estimates

The purpose of this overview is to provide estimates of carbon burial and sequestration to estimate the stock and flow of climate regulation services provided by coastal habitats in the UK.

In 1999, Cannell *et al.* published a paper describing the process and findings of a comprehensive national inventory of CO_2-carbon terrestrial sinks and sources for the UK. For **salt marshes**, an average carbon sequestration rate in British salt marshes of 1.4tC/ha/yr is given, varying from 0.64tC/ha/yr in north Norfolk to 2.19tC/ha/yr in the Solent. These figures are based on estimates of the annual rate of sediment accretion, sediment bulk density and carbon content of these sediments.

There are a number of reviews and they often use the same primary studies. Duarte *et al.* (2005) review the literature on organic carbon burial and provide the following estimates (in tC/ha/yr): 1.39 in **mangroves**, 1.51 in **salt marshes**, 0.83 in **seagrasses**, 0.045 in estuaries and 0.17 in the coastal shelf. [43]

An updated review of five studies on carbon sequestration in **seagrasses** (species specific) is provided in Duarte *et al.* (2011), all based on Mediterranean data. This study provides a figure of 0.53tC/ha/yr for short-term (years) carbon storage in seagrass sediments, and a longer term rate of 0.58tC/ha/yr. Estimates vary between 0.029tC/ha/yr for *Zostera noltii* to 1.98tC/ha/yr for *P. Oceanica*.[44] The former species occurs in the UK. *Zostera marina* also occurs in the UK and has a burial rate of 0.524tC/ha/yr in the Mediterranean (based on Cebrian *et al.,* 1997).

McLeod *et al.* (2011) provide another literature review, using estimates from Chmura *et al.* (2003), Duarte *et al.* (2005), Bird *et al.* (2004), Kennedy *et al.* (2010), Lovelock *et al.* (2010) and Sanders *et al.* (2010). They provide the following average estimates: 2.26 ±0.39tC/ha/yr for **salt marshes**, 0.57±0.06 for **intertidal marshes**, and 1.38±0.38 for **seagrass beds**.

Some of the original studies are included here. Chmura *et al.* (2003) analyse a large dataset on carbon sequestration in **mangroves** and **salt marshes**. They find an average rate of carbon sequestration of 0.573tC/ha/yr, with no significant difference between mangrove and salt marshes.
Bird *et al.* (2004) study carbon burial in Holocene **mangroves** in Singapore. They show that carbon sequestration rates ranged from 0.9 to 1.7tC/ha/yr. The lower estimate applies to mangrove mud, whereas the higher estimate applies to mangrove peat. They compare these estimates to a study by Fujimoto (2000) that found rates calculated on centennial to millennial timescales for mangroves in the Asia-Pacific region between 0.14 and 2.98tC/ha/yr.

Sanders *et al.* (2010) study a mangrove forest in Brazil with mangrove margins and intertidal mudflats. Burial rates were estimated to be 11.29tC/ha/yr for the **mud flat**, 9.49tC/ha/yr for the **margin** and 3.53tC/ha/yr for the **mangrove forest**.

Lovelock *et al.* (2010) estimate carbon accumulation in **mangroves** in New Zealand and provide an estimate of 3.67tC/ha/yr. Along the Great Barrier Reef in Australia, Brunskill *et al.* (2002) studied organic carbon burial rates in a **mangrove** system near a river mouth. The highest rate is found within shallow

[43] For references, see Duarte *et al.* (2005).
[44] For references, see Duarte *et al.* (2011).

(<20m) wind-protected embayment. Organic carbon burial rates average was found to be 15molC/m2/yr (1.8tC/ha/yr) in mangroves between 0 and 5 m water depth, and 1.7 (0.2tC/ha/yr) in areas 5-20m below sea level.[45] These areas served as deposition locations for riverine sediments and also receive mangrove carbon. Beyond 20m the rate drops to less than 0.0025mol/m2/yr (~0tC/ha/yr). The tropical **coral reef** area has an OC burial rate of 0.8molOC/m2/yr (0.0096tC/ha/yr).

Kennedy *et al.* (2010) analyse data from 207 **seagrass** sites at 88 locations around the globe. Their results indicate an accumulation of seagrass organic matter of between 0.41 and 0.66tC/ha/yr and a similar contribution from allochthonous organic carbon. They assume, based on their data, that half of the organic C buried in seagrass sediments derives from the seagrass tissue, whilst the other half comes from trapping of other particles, as the seagrass canopies facilitate sedimentation and reduce resuspension.

[45] Conversion rate: 1 molC/m2 = 120 kgC/ha.

Appendix 3.10.7 Importance of marine ecosystem services

To develop a matrix that indicates the importance of marine ecosystem services, we organised an expert meeting. Nine coastal and marine experts were asked to score the importance for human wellbeing of the final ecosystem services for each habitat with three different signs, reflecting 'high', 'medium' or 'no' importance.

The expert scores were given numerical values of 10, 5 and 0 respectively. Non responses were left blank. The mean and median of these scores were calculated. Mean scores were assigned back to importance levels as follows: low for 0-2.9, medium for 3.0-6.9, and high for 7.0-10. Where means and medians did not coincide, the table gives the median in brackets.

The results are presented in **Table 4.A4.** It should be noted that this is a nation-wide assessment which may average out sites that are of particular local importance.

Table 4.A4. Importance of ecosystem services per marine habitat.

	Products *	Sea defence	Erosion prevention	Healthy climate†	Tourism and nature watching	Education research	Aesthetic: property	Spiritual, aesthetic: wild species, seascapes
Mudflats	medium	High (medium)	medium	high	medium	medium	medium	high
Mangroves	high	high	high	high	medium	medium	low	high
Inter. wetland	medium	high	high	high	medium	medium	medium	high
Seagrass beds	medium	low	medium	medium	medium	medium	low	high
Kelp forest	medium	low	low	medium	medium	medium	low	high
Estuaries	high	medium	low	medium	high	high	high	high
Coral reefs	high	low	low	medium	low	high	low	high
Rocky bottom	medium	low	low	low	medium	high	medium	high
Coastal shelf	high	low	low	medium	medium	medium	low	medium
Open ocean	high	low	low	medium	low	medium	low	medium

* Experts were explicitly instructed to consider the nursery function.

† Experts commented that carbon sequestration is of high importance for the outer coastal shelf, but of medium importance for the inner shelf. For the open ocean, sequestration rates depend much on location. Sequestration in seagrass beds and kelp forests is of importance within a time frame of decades, yet otherwise not important.