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AN INTRODUCTION TO SOCIO-ECONOMIC ASSESSMENT WITHIN A MARINE STRATEGY FRAMEWORK

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Introduction

1.1 Purpose of the report

The purpose of this report is to set out a decision support system focused on socio-economic analysis aimed at enabling OSPAR to further its mission – ‘to conserve marine ecosystems and safeguard human health in the North-East Atlantic by preventing and eliminating pollution...’. The socio-economic methods and techniques set out in this report can contribute to the protection of the marine environment and the sustainable use of the seas by identifying economically efficient and cost effective policy options (i.e. projects, policies, programmes and courses of action). The socio-economic decision support system (DSS) can be embedded into the OSPAR work programme and its core holistic assessment of the quality status of the N-E Atlantic and its future prospects. The particular version of socio-economic analysis advocated in this report has been deliberately chosen because of its compatibility with the ecosystem approach (EA) adopted by OSPAR in line with the Ministerial Declarations and statements which have guided the Commission’s work since 1998. OSPAR defines EA as: “the comprehensive integrated management of human activities based on the best available scientific knowledge about the ecosystem and its dynamics, in order to identify and take action on influences which are critical to the health of marine ecosystems, thereby achieving sustainable use of ecosystem goods and services and maintenance of ecosystem integrity”. The socio-economic analysis can underpin the on-going work on contaminants abatement measures and quality assessment and risk indicators science that OSPAR has championed. It takes a strong sustainability position in that ecosystems are seen as suppliers of a range of intermediate and final services through which humans benefit in terms of welfare. Sustainable utilisation of this vital resource base is therefore the key notion. The assigning of monetary values to the benefits provided by ‘healthy’ ecosystems it is argued can supplement scientific and ethical arguments in favour of environmental protection and biodiversity conservation.

The EU Marine Strategy Framework Directive (MSFD) 2008/56/EC presents a further set of challenges in its setting out of community action relating to marine environmental policy. Article 8.1 (c) calls for ‘an economic and social analysis of the use of those waters and of the cost of degradation of the marine environment’. Both the MSFD and the Common Fisheries Policy are ‘informed’ by the EA, with Good Environmental Status interpreted in terms of ecosystem functioning and services provision. Implementation of the EA should be via so-called adaptive management policy and practice. This is essentially ‘learning by doing’ approach with policy and practice being constantly monitored and re-orientated/changed as experience is gained during implementation. Such an approach accepts the inherent complexities and uncertainties that often shroud the utilisation of marine resources. Problems of resource overexploitation and/or environmental quality degradation tend to have multiple causes and are evolutionary. The adaptive

management process will be composed of a number of sequential but overlapping components:

- baseline science to inform the management process in terms of the ecosystem structure, process and forcing vectors that condition the coevolving socio-economic and ecological marine system and its inherent trends;
- the application of methods and techniques (the toolbox) for the assessment of the marine system's status and future prospects;
- focused analysis of contemporary 'key' and potentially significant emerging issues due to overarching environmental change;
- participatory and deliberative methods and techniques to foster social dialogue amongst all relevant interest groups, and to search for 'values' consensus/majority positions;
- modelling to compare alternative policy option outcomes;
- development of appropriate indicators; and
- adequate monitoring and review procedures.

The DSS detailed in this report represents one component of the adaptive management approach and can form the basis for the type of analysis called for under the Marine Strategy Directive. The term 'socio-economic analysis' is open to a wide range of interpretation and Annex 4 in this report outlines the approaches that a number of OSPAR countries have or are in the process of undertaking. While all these efforts will usefully augment the knowledge base relating to human activities and the financial impacts associated with marine resource usage, only a minority go further and investigate the full economic implications. The distinction between a narrowly-based financial analysis of activities such as, for example, fishing and shipping and a more comprehensive ecological economic analysis is important and Section 4 of this report focuses on this and related issues (market and non-market related values). A set of studies concentrating on the eutrophication of the Baltic and future possible quality states (related to a number of costed abatement options) is a good exemplar for future OSPAR initiatives. The Baltic studies compare the economic costs and benefits of eutrophication abatement at a drainage basin scale (see Annex 4). These studies also serve to highlight the fact that socio-economic analysis will need to be capable of handling coastal/marine relative change management issues from the local to the international spatial scale and over temporal scales up to at least 2020.

The possible relative changes in quality status and the human-related activities which serve to pressurize the marine environment can be modelled within an appropriately designed DSS. In this report an initial scoping stage is recommended based on the so-called DP-S-I-R framework (Driving Pressures – State Changes – Impacts – Policy Responses). The DP-S-I-R scoping work would facilitate cooperation across OSPAR (feeding also into EU policy actions) and could lead to the prioritising of ecosystem change sequences linked to human activities on a regional sea/catchment scale. More locally focused assessments on any given environmental problem/policy response would also not be precluded in this DSS. While it is the case that marine system issues can be complex and that a range/combination of variables

influence human interest groups under any given governance system, partial decomposition of problems is possible (Ostrom, 2007). According to Ostrom (2007) a particular social-ecological system (SES) is essentially a resource system (e.g. a fishery or a more complex estuary-based set of ecosystems and human activities) which yields valued resource outputs, plus a set of resource users and a prevailing governance system. The SES itself is embedded in a larger socio-economic, political and ecological context (increasingly global in scale).

The temporal scale of the environmental changes can be highlighted and 'modelled' via so-called scenario analysis. 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 relevant policy decision information on feasible future states of the world. It is vitally important that the futures scenarios chosen should be consistent across all OSPAR members and, for optimal effect, in line with EU and IPCC etc practices.

Given the sequential format of the DSS recommended in this report, it is possible to discern a future work programme up to 2012. Both the DP-S-I-R scoping assessment and the scenario-based analysis could feasibly be completed in that timescale through a cooperative effort involving all Member States. A comprehensive economic assessment of all the relevant gains and losses (costs and benefits) associated with the baseline and other change scenarios is a more difficult goal if new original economic valuation studies are required. While a number of valuation studies exist for European marine and coastal ecosystems, the database is not currently sufficient for a full in-depth assessment of marine areas/drainage basins at the scale of, for example, the North Sea. An initial economic valuation exercise would, however, serve to identify data gaps and could set the foundations for a more spatially extensive and comprehensive analysis. Individual country studies and/or studies of local or individual problem issues would augment the general stock of relevant knowledge.

1.2 Economic assessment in a systems approach

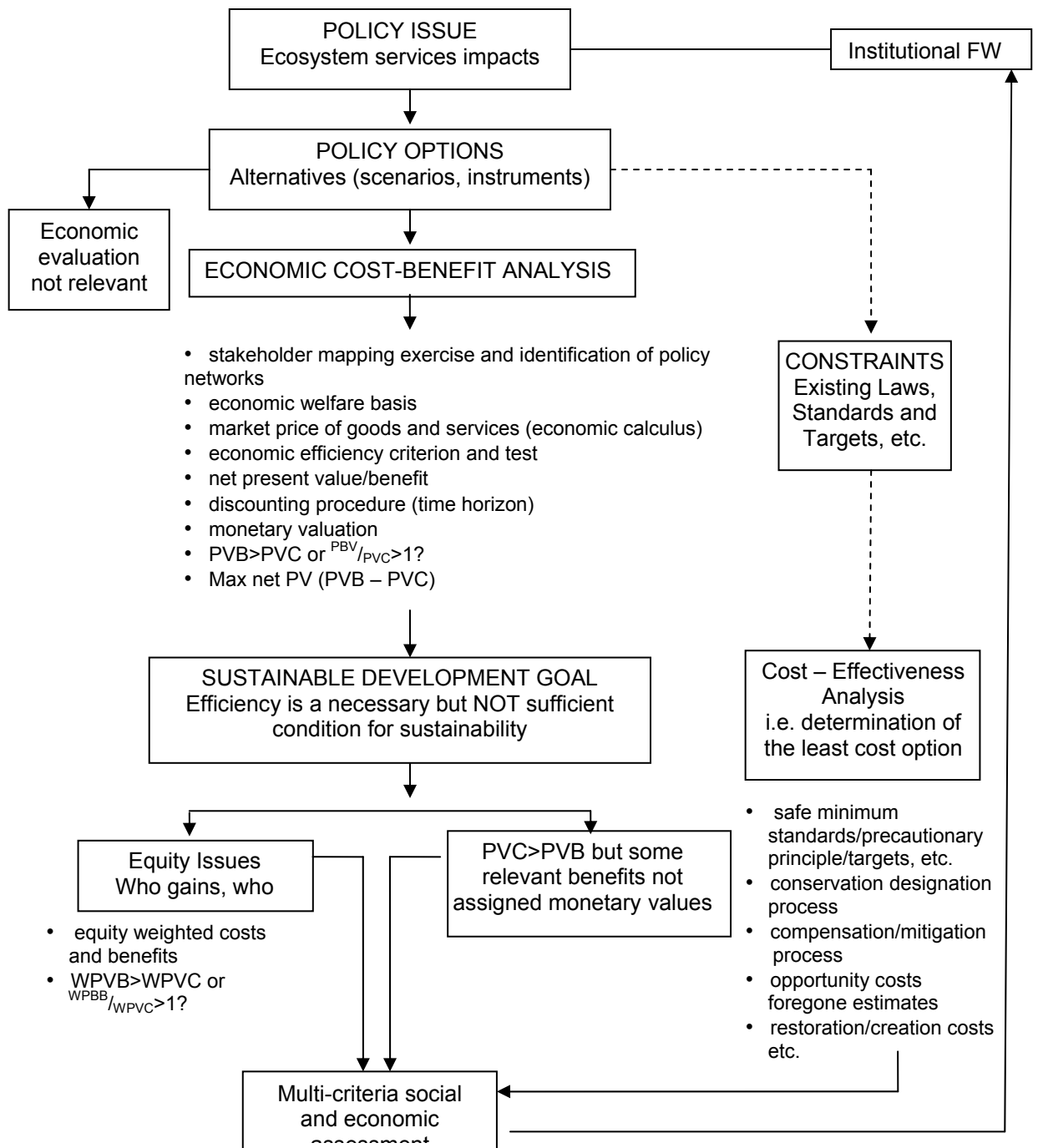
The process of economic assessment can only take place after policy issues have been identified within given spatial and temporal scales and scenarios and evaluative criteria have been established. Once agreed, the policy issues and scenarios that are identified by this process then provide the framework (socio-economic assessment) within which the economic assessment can be constructed. Note however that this is not a one-way process. Ideally, feedback should occur between all stages of the assessment process and the deliberative systems set up with stakeholders, since questions that are thrown up by the assessment can help to refine the policy issues and scenarios that are of concern to stakeholders. Most problems situations involve competing uses for marine resources and are conditioned by the governance that is in place.

The economic assessment of a marine or coastal zone policy issue or set of issues must be underpinned by biophysical research and data relating to the various ecosystem processes, structures, stocks, flows and dose response relationships (e.g. quantified relationships between pollutant discharges and related environmental state changes). Marine and related (catchment and coastal) ecosystems provide a range of service outcomes, many of which are deemed valuable goods ('benefits') by human society. So ecosystem final services and goods are the aspects of ecosystems consumed and/or utilized to produce human well-being. They result from complex interactions that occur at multiple spatial and temporal scales. Ecosystem processes can have indirect and direct outcomes leading to welfare gains and sometimes losses (benefits) and single processes often yield multiple services (joint products). Because of these characteristics a systems approach is necessary, in understanding all of the links, if monitoring, measuring and valuing ecosystem services is to be done in a meaningful way. The ultimate goal is to ensure a relatively sustainable and productive utilisation of the available resource systems and the avoidance of irreversible system changes/collapse with consequential high welfare losses.

The resource system policy issues under investigation will be composed of a complex mixture of environmental and socio-political driving processes, consequent environmental state changes which then impact on the provision of ecosystem services, goods, and human welfare. The distribution of the welfare gains and losses in society, together with existing policy measures and networks will influence policy response strategies. The economic analysis (Cost benefit analysis (CBA) and Cost effectiveness analysis (CEA)) seeks to evaluate the social welfare gains and losses involved from an economic efficiency perspective, tempered by any relevant distributional equity considerations, other precautionary environmental standards and regional economic constraints (most often focused on 'local' employment and economic multiplier impacts which can result in cultural and community losses or gains) – see Figure 1.

The policy issues that are relevant in any particular management context can be identified via a scoping process, facilitated by the so-called DP-S-I-R framework.

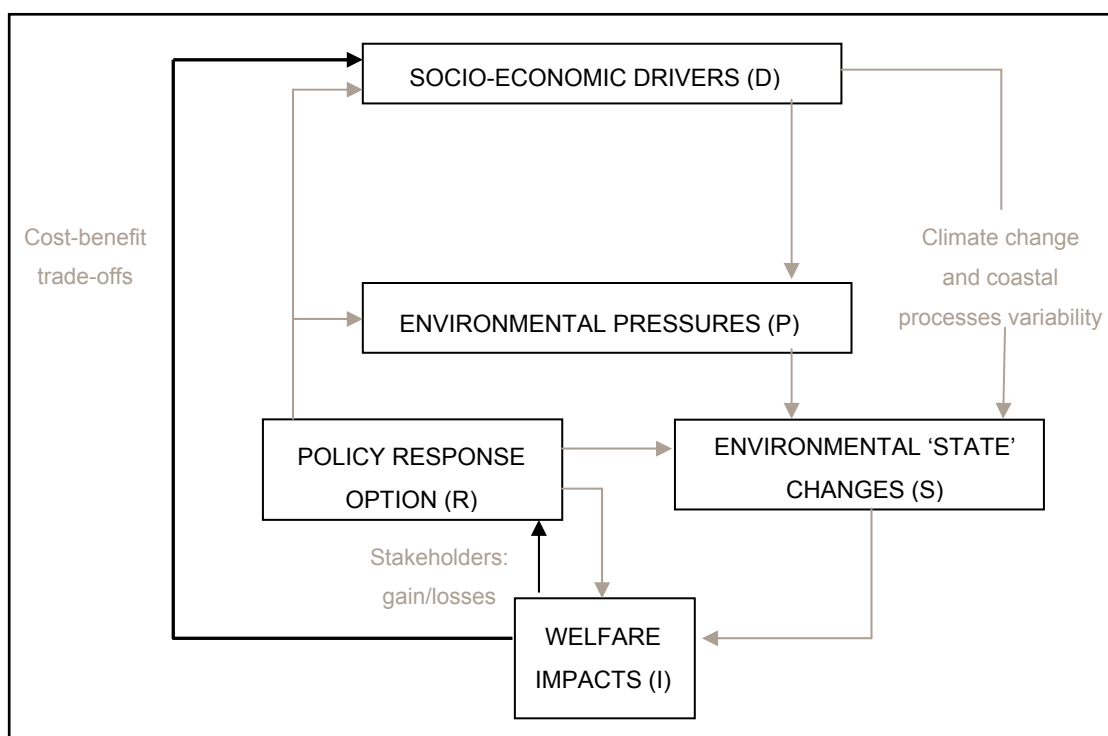
Figure 1 Overview of the process of economic assessment



2 The DPSIR analytical Framework

The DPSIR framework (see Figure 2) is a useful device for clarifying the role that socio-economic drivers play in inducing pressures on the environment (over varying timescales and across a range of spatial scales). These pressures result in state changes (often ecosystems degradation or loss) and consequent impacts on the welfare of people and communities locally, regionally and sometime globally. Efforts to modify the impacts (policy responses) produce feedback effects within the drivers/pressures systems (Turner et al., 1998).

Figure 2 DPSIR, adapted from Turner et al (1998)



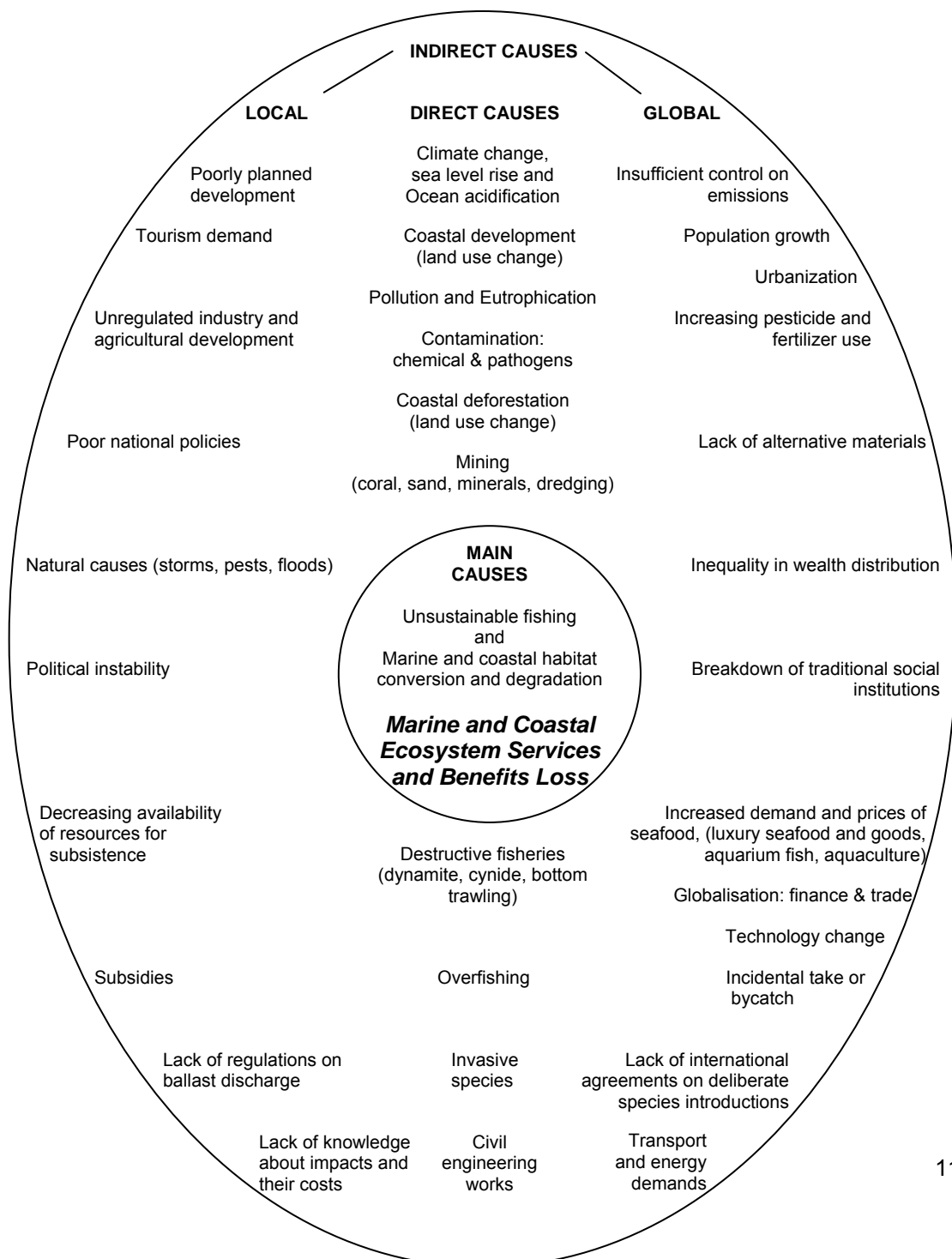
Coastal zone and regional seas management, for example, is hindered by, among other factors, the scale mismatch problem which has intensified as the process of globalisation has itself accelerated. Coastal zone issues are often conditioned by an historical legacy e.g. the build up of contaminants in estuarine and coastal sediments from past industrial/urban development; or chronic eutrophication from intensive agriculture and/or inadequate sewage treatment facilities etc. This (negative) legacy impact on ecosystem services provision can be difficult and costly to ameliorate e.g. cleaning sediments or aquifers or modifying coastal defence structures.

The socio-economic drivers of environmental change in marine and coastal zones are increasingly regional and global in scale and the local population may have little leverage over them. The vulnerability of marine and coastal ecosystems is increasing because of a combination of exposure to natural (often weather and climate change related) events, storms etc. and the workings of the global economy and its deregulated components (industrial

production/location, trade, advertising etc.). A further element in the increased vulnerability equation is the growth in world population and the fact that the more hospitable coastal zones are already densely populated with little future margin for out-migration.

Figure 3 illustrates the type of drivers and pressures which are most relevant to marine and coastal zones and their ecosystems. The framework highlights the direct and indirect causal factors of environmental change and also the need to clarify the juxtaposition of temporal and spatial scales involved. The key direct drivers for coastal zones seem to be land use change and habitat loss and climate change; and indirect drivers such as shifting consumer preferences and diets (particularly in richer countries), population growth and globalisation in terms of finance and trade arrangements (MEA, 2005).

Figure 3 Drivers and pressures on marine and coastal ecosystem services and benefits



2.1 Scaling mismatch and globalisation

Some of ecosystem loss and degradation problems are confined more or less to the local scale (i.e. within coastal zones). The drivers and pressures and their impacts are in these instances, at least in principle, open to local management actions. But problems such as eutrophication of estuaries and coastal waters have to be viewed at the regional sea/catchment scale. The drivers and pressures, agricultural intensification/expansion etc., are located in physical catchments or political designations which extend well beyond the coastal zone. Increasingly, the drivers of change are very distantly located from the ecological impacts and consequent socio-economic cost effects. A combination of globalised elements, remote markets, heavily advertised goods and services which condition consumer preferences, financial markets, trade arrangements, transport networks, regulatory regimes (or the lack of regimes) and international labour cost differentials, all contribute to ecosystem loss in marine and coastal zones. The global economy and engine of economic growth, international trade, is characterised by a focus on short term financial returns, 'light touch' regulation of markets and trading arrangements and an underlying growth imperative measured in terms of GDP/GNP maximisation rather than qualitative development progress. The model appears to assume that economic activity can expand indefinitely without regard for either source or sink environmental limits.

By way of illustration this section outlines an application of the DP-S-I components of the overall scoping method (DP-S-I-R) to the OSPAR area.

The North-East Atlantic is characterised by a variety of physical and biological characteristics, ranging from open ocean to shallow coastal waters. Within the N-E Atlantic, the North Sea is a semi-enclosed area, significantly impacted by socio-economic activities. Over 160 million people live in the North Sea catchment, which also hosts large industrial/urban agglomerations and has been subjected to extensive land use change. Despite significant improvements in pollution abatement policy and practice, coastal waters are still threatened by contamination risks (e.g. endocrine disruption on marine organisms). Dredging and disposal to maintain essential navigation or to extend/create ports can disturb contaminants that have accumulated over long periods of time in estuarine and other sediments (historical legacy problem). This region is also one of the most heavily utilised shipping routes with an attendant risk of alien species introduction. Finally, the North Sea has been directly impacted by resource exploitation activities such as fishing, transport, tourism, oil and gas extraction, sand and gravel extraction and most recently off shore wind farming.

Fishing pressure is the most widespread in the N-E Atlantic region, with demersal fish stock (plaice, cod, haddock) coming under significant strain. In addition, fishing affects submarine habitats and seabird populations.

Eutrophication is also a major concern in some regions of the N-E Atlantic, although it is restricted to semi-enclosed seas and coastal waters, such as the Southern North Sea. **Climate change** is likely to increase water temperature which may exacerbate the eutrophication problem; and also differentially

impact warm water and cold water species. Any impact on lower trophic levels will influence the success of target fisheries species. More winter precipitation and more episodic flooding events will have a direct impact on economic activity and related physical assets, as well as altering the fluxes of nutrients and chemicals entering coastal waters. Sea level rise and acidification are among other adverse impacts.

Shipping activity has increased significantly in the N-E Atlantic, with larger vessels requiring new port facilities and increased dredging of navigation channels. These activities have negative impacts such as, for example, loss of habitat, redistribution of contaminated sediments, introduction of alien species via ballast water, physical effects on coastlines via accidents. These so-called negative externalities need to be set against the financial benefits provided by shipping/trading activities .

Chemical pollution is also a concern in the N-E Atlantic. A wide range of contaminants exist in the marine environment, many of which are shrouded in uncertainty in terms of their damage functions. The generic decline (due to better regulation) in riverine concentration of most metals and organic contaminants has not yet had a clear cut effect on the biota due to variations in bioavailability, local conditions and a legacy of past contamination in sediments. Other synthetic chemical releases have been linked to endocrine disruption affects.

The N-E Atlantic is a prime site for **renewable energy installations** such as offshore windfarms, energy wave devices and tidal barriers. These schemes all have local environmental impacts and in the case of windfarms a regional scale 'displacement' impact e.g. displacement of fishing by marine protected areas around wind turbine sites and consequent increase fishing pressure in 'unprotected' areas. Aquaculture also continues to expand with local environmental consequences and potential impacts on the marine food web via fish food provision and accidental releases of fish with a low genetic diversity (Langmead et al., 2007) – See Figure 4.

Figure 4 Scenario outcomes for the North-East Atlantic

		Current Trend	Baseline Scenario	Scenarios (relative to current)			
				National Enterprise	Local Responsibility	World Markets	Global Community
DRIVERS	UWWT	↗	↗	↘	↗	↘	↗
	Industrial discharge	↘	↘	↘	↘	↘	↘
	Fossil fuel en. gen.	↘	↗	↗	↘	↗	↘
	N fertiliser use	↘	↘	↗	↘	↗	↘
	P fertiliser use	↘	↘	↘	↘	↘	↘
	Livestock prod.	↗	↗	↘	↘	↘	↘
	Dredg. & spoil disp.	↗	↗	↘	↘	↗	↗
	Dredg. & trawl effort	↘	↘	↗	↘	↘	↘
	Fishing effort	↘	↘	↘	↘	↗	↘
PRESSURES	[PCB] seawater	↘	↘	↘	↘	↘	↘
	[Cd] seawater	↘	↘	↗	↘	↗	↘
	Total nitrogen	↘	↘	↗	↘	↗	↘
	Total phosphorus	↘	↘	↘	↘	↘	↘
	TSS	↘	↘	↗	↘	↘	↘
	Demersal catch	↘	↘	↘	↘	↘	↘
	Pelagic catch*	↘	↘	↘	↗	↘	↘
STATES	[PCB] mussels	↘	↘	↘	↘	↘	↘
	[Cd] mussels	↘	↘	↗	↘	↗	↘
	Eutrophication	↗	↗	↘	↗	↘	↗
	TWB status	↗	↗	↘	↗	↗	↗
	Waterfowl abund.	↘	↘	↘	↘	↘	↘
	Submarine sediment hab.	↘	↘	↘	↗	↘	↘
	Demersal stocks	↘	↗	↘	↗	↗	↗
	Seabird abundance	↘	↘	↘	↗	↘	↘
	Pelagic stocks*	↘	↘	↘	↗	↘	↘

*trend for sandeels is ↗ and herring ↘

Source: Langmead et al. (2007).

Putting an economic value on all these impacts is a complicated and as yet only partial possibility. The methods and techniques available for such an exercise are detailed in later sections of this report. Before these sections, we now take a closer look at the use of scenario analysis.

3 Scenario analysis

A scenario can be defined as a coherent, internally consistent and plausible description of a possible future state of the world (Parry, 2003). It needs to be emphasised that a scenario is not a forecast because it cannot assign probabilities to any particular outcome. Instead, scenarios portray images of how society and its supporting environment could look like given different sets of assumptions and consequent conditions. Scenarios typically contain qualitative storylines augmented by varying amounts of quantified data. They can be informed by relevant history but not conditioned by it, except in the case of so-called baseline or 'business as usual' (BAU) scenarios. The latter can be utilised as benchmarks against which to portray other possible states of the world and are completed with the aid of trend data. Table 1 presents a simple typology of scenarios characteristics in terms of basic principles. In practice, scenarios will combine a range of features depending on their real world application and the scale at which they are pitched.

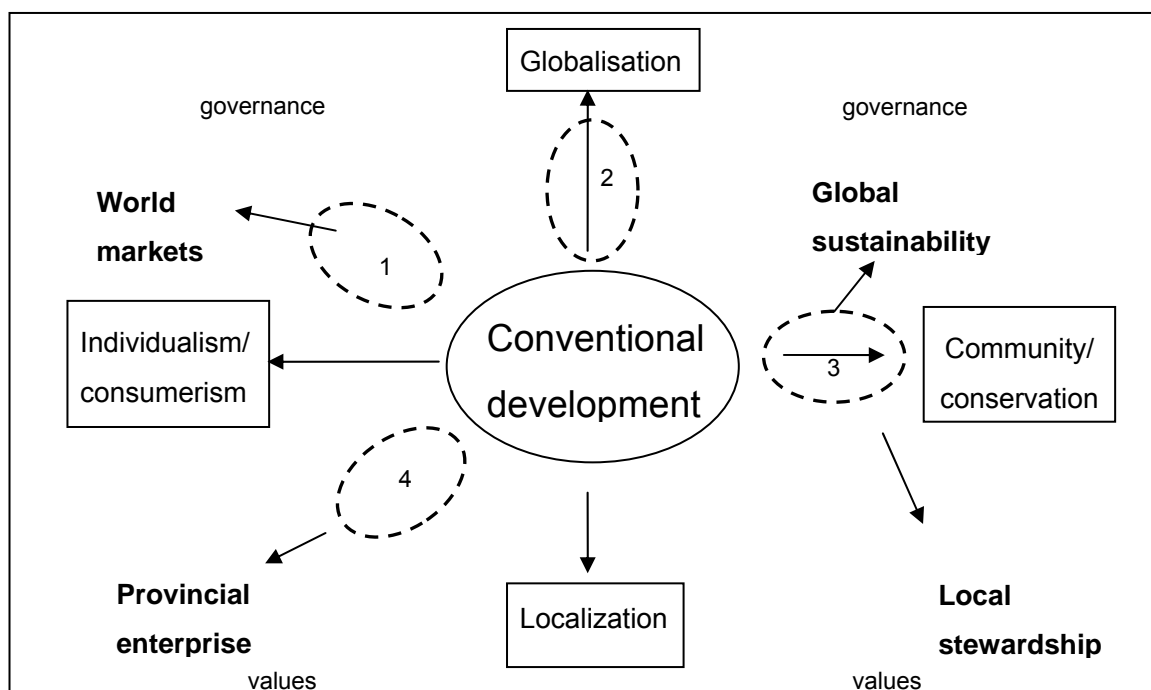
Table 1 Scenario characteristics typology (adapted from EEA, 2000)

Type 1	Forecasting scenarios: they attempt to encompass future alternative development paths from the standpoint of the current situation (time = t_0); they can also include expected of desired policy switches.
Type 2	Backcasting scenarios: they take as their initial start point some desired future (time = t_{1-n}) state of affairs or policy objective and then explore alternative strategies to maximise goal attainment.
Type 3	Descriptive scenarios: they set out a sequenced set of possible events in a neutral way.
Type 4	Normative scenarios: their sequences explicitly incorporate different interests, values and ethics.
Type 5	Quantitative scenarios: usually computable model-based exercises.
Type 6	Qualitative scenarios: which rely solely on narratives.
Type 7	Trend (BAU) scenarios: based on the extrapolation of current trends.
Type 8	Peripheral scenarios: which attempt to include surprises, i.e. unlikely and/or extreme events and their consequences.

Scenarios can aid decision makers in their efforts to cope with inevitable uncertainty, over temporal scales typically ranging from 10 to 100 years. They can also be used to facilitate consensus or negotiation in situations where multiple competing stakeholder interests are at issue. They can be focused on particular policy objectives and/or instruments and provide sensitivity assessments. Alternative scenario visions are most often reflected against a baseline (BAU) trend scenario.

There is no shortage of candidate scenarios to choose from and to adapt to the N-E Atlantic region. In the sub-section that follows a hybrid approach is outlined which borrows from a set of scenarios previously formulated to investigate the impact of climate change, technological advances and environmental consequences in a range of contexts - see Figure 4.

Figure 5 Environmental Future Scenarios (to 2080)



Notes: the four quadrants are separated by 'zones of transition' not distinct boundaries.

Source: OST/DTI, Environment Futures, Foresight, OST London

<http://www.foresight.gov.uk>. See also Lorenzoni et al. (2000); and UNEP (2002).

The aim is to first provide a set of basic contextual narratives within which to set four somewhat more specific scenarios (UNEP, 2002) with relevance for sea-coastal-catchment areas across the European sub-regions (Western, Central and Eastern). The critical issues, which are highlighted, mirror the pressures, impacts and responses within the DP-S-I-R framework.

The narrative contexts and the scenarios are framed by two orthogonal axes, representing characteristics grouped around the concepts of societal values and forms of governance. The value axis provides a spectrum from individualistic, self-interested, consumer/market-based preferences, through to collectivist, citizen-based communitarian preferences often with a conservationist bias. The vertical axis spans levels of effective governance from local to global. The four quadrants are not sharply differentiated but rather are bounded by overlapping transitional zones not distinct boundaries. Change occurs as certain trends and characteristics become more or less dominant across the different spheres of modern life-government, business, social, cultural and environmental – see Figure 5.

Taking the four contextual backgrounds conditions first, *World Markets* is dominated by globalisation, which fosters technocentric and often short-term societal views. Expectations about an expanding EU and Single Market are born out and economic growth remains the prime policy objective. Environmental concerns are assumed to be tackled by a combination of market-incentive measures, voluntary agreements between business and government and technological innovation. Decoupling of the growth process

from environmental degradation is assumed to be feasible, not least because ecosystems are often resilient. Weak sustainability thinking is favoured and 'no-regret' and 'win-win' options are the only ones pushed hard by regulators. Rapid technological change, sometimes unplanned, will be the norm, as will trade and population migration. Private healthcare, information technology, biotechnology and pharmaceutical sectors of the economy, for example, will thrive, while 'sunset' industries will rapidly disintegrate (e.g. heavy engineering, mining and some basic manufacturing). The internal and external boundaries of state will retreat producing a more hollowed out structure (Jordan et al., 2000). National governments will struggle to impose macroeconomic controls as transnational corporate power and influence escalates. Multilateral environmental agreements will prove problematic and prone to enforcement failures.

Under *Global Sustainability*, there would be a strong emphasis on international/global agreements and solutions. The process would be by and large 'top down' governance. Trade and population migration would still increase but within limits often tempered by environmental considerations. EU expansions would be realised but social inequities would receive specific policy attention via technology transfer, financial compensation and debt for nature swaps/agri-environmental programmes and other PES's (payment for ecosystem services schemes).

Provisional Enterprise would be a much more heterogeneous world, EU expansion might stall and a slow process of fragmentation (economically and politically) might be fostered. A protectionist mentality would prove popular and economic growth, trade and international agreement making prospects would all suffer.

Local Stewardship would put environmental conservation (ecocentrism) as a high priority. A very strong sustainability strategy would be seen as the only long term option. This strategy would emphasise the need for a reorientation of society's values and forms of governance, down to the local community scale. Decentralisation of economic and social systems would be enforced, so that over time local needs and circumstances become the prime focus for policy. Economic growth, trade, tourism (international) and population migration trends would be slowed and in some cases reversed.

With this backdrop in mind, the four scenarios 1, 2, 3, and 4 can be roughly located in Figure 4, with the arrows indicating the general direction of change over decadal time. Scenario 1 is almost a trend/baseline scenario. The policy goal of maximise GDP growth is achieved via an extended single market system stretching into central Asia. New accession states are given transitional status to ease their progress into market-based systems. The relatively weak enforcement of environmental standards in these countries fosters short run profitability but may hinder long run resource use efficiencies. Rapidly growing volumes of trade and travel increase the level of economic interdependence in Europe, but social cohesion remains somewhat weak, as people strive to satisfy individual consumerist preferences. Scenario 2 imposes sustainability constraints via a 'top down' governance process but

also encourages citizens to 'think global and act local'. Scenario 3 allows for a much more radical paradigm shift in societal values and organisations, environmental conservation and social equity rise up the political priority agenda. In Scenario 4, protectionism breeds growing disparities across the sub-regions of Europe. Inequality and possible conflicts spawn a relative isolationist response at the nation state level. So now we turn to the implications for the future regional seas/coastal zones in Europe, given the different scenarios.

Under all scenarios, condition pressures on coastal ecosystems are seen to increase, either through direct exploitation of marine and coastal resources, including local use changes and an increase in the built environment at the coast; or through changes in related catchments associated with the spatial planning of development and transportation policies, changes in agriculture policy, especially trade regimes and reform of the Common Agricultural Policy (CAP). Another feature of the scenario analysis is that the impact of climate change does not vary significantly across scenarios until around 2030 – 2050 because of delays in the response of the climate system. But uncertainty is a particular problem in this context and climate impact predictions are changing rapidly.

Focusing on the N-E Atlantic region more specifically, under the BAU scenario a reduction in nutrients inputs continues but eutrophication probably still increases, due to climatic effects. Implementation of the Common Fisheries Policy results in some improvements in demersal fish stocks. The North Sea transitional water bodies status increases, partly due to reductions in contaminant and nutrient loading and this triggers a fall in waterfowl abundance. Under both the 'Global Sustainability' and 'Local Stewardship' scenarios, fisheries are more sustainable and some stocks return to previous levels. There is a mixed impact on seabirds, with divers increasing and discards feeders declining. The spread of marine protected areas safeguards biogenic reefs and other rare habitats. Stricter rules on fishing gear selectively reduce the damage impact on benthic communities. Under 'World Markets' and 'Provincial Enterprise' conditions, fisheries will be unsustainably exploited with negative effects on seabirds, non-target species and submarine sediment habitats. Increased transport/shipping activity stimulates more dredging, increases the risk of alien species introduction and contamination of the water column (Langmead et al. 2007) – see Figure 4.

An alternative approach to scenario analysis would involve the comparison of an agreed BAU baseline scenario and outcomes against one or more scenarios which change the baseline through the introduction of a set of policy measures. The implications of switching various policy measures 'on' and 'off' could then be assessed.

4 Financial versus Economic Valuation

4.1 Prices versus Values

In any socio-economic assessment it is necessary to distinguish between financial and economic values and analysis. Prices and values are not necessarily equivalent and in fact price is only that portion of the underlying value of a good which is realised in the market place. 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/subsidy etc interventions. Prices will typically diverge from values when so-called public goods (non-exclusion non-rivalness 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 prices. 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 'willingness to pay' for changes in the provision of the good.

Note that this measure is not the same thing as actual payment; when the latter is less than the former a consumer gains value (consumer surplus). A number of methods have been developed to estimate the value of any good. They range from adjusted market prices, through productivity effect methods and revealed preference (based on consumer actions) to survey-based expressed preference methods. Market prices, for example, can be used to estimate part of the value of improved water quality by quantifying the increased value of commercial catches. See Annex 4 in this report for some fisheries analysis.

Economics in the process of systems assessment

Valuation in economics theory and practice has often been approached in terms of 'opportunity cost'. This means that the value of an environmental asset or service (or a damage avoided) is assessed in terms of the 'tradeoffs' associated with obtaining or maintaining that good. Some approaches further attempt to quantify these costs in monetary terms by identifying a trade-off between the selected environmental benefit (asset, service, or damage avoided) and economic goods and services for which price-tags are already attached. If this approach is pursued comprehensively, it becomes possible in principle to compare all economic and environmental goods and services (and damages) in monetary terms, and to look for 'highest value' uses of economic and environmental resources in these terms.

The scheme below summarises the initial analytical step which is to decide on whether environmental evaluation seeking to determine opportunity costs for a project, policy or programme of marine and/or coastal intervention, or a more constrained cost-effectiveness analysis is required (see Figure 1). In the latter context, a range of options are usually assessed to see which yields the desired outcome, e.g. achievement of a given water quality status, 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 (see Annex 1 for more details).

- Step 1: Identifying policy options
- Step 2: Characterise (describe in economic terms) the options
- Step 3: Provide economic assessment component
- Step 4: Overall systems assessment using multi-criteria approach
- Back to 1 (iterative process)

The policy response **interventions** usually fall into a number of categories:

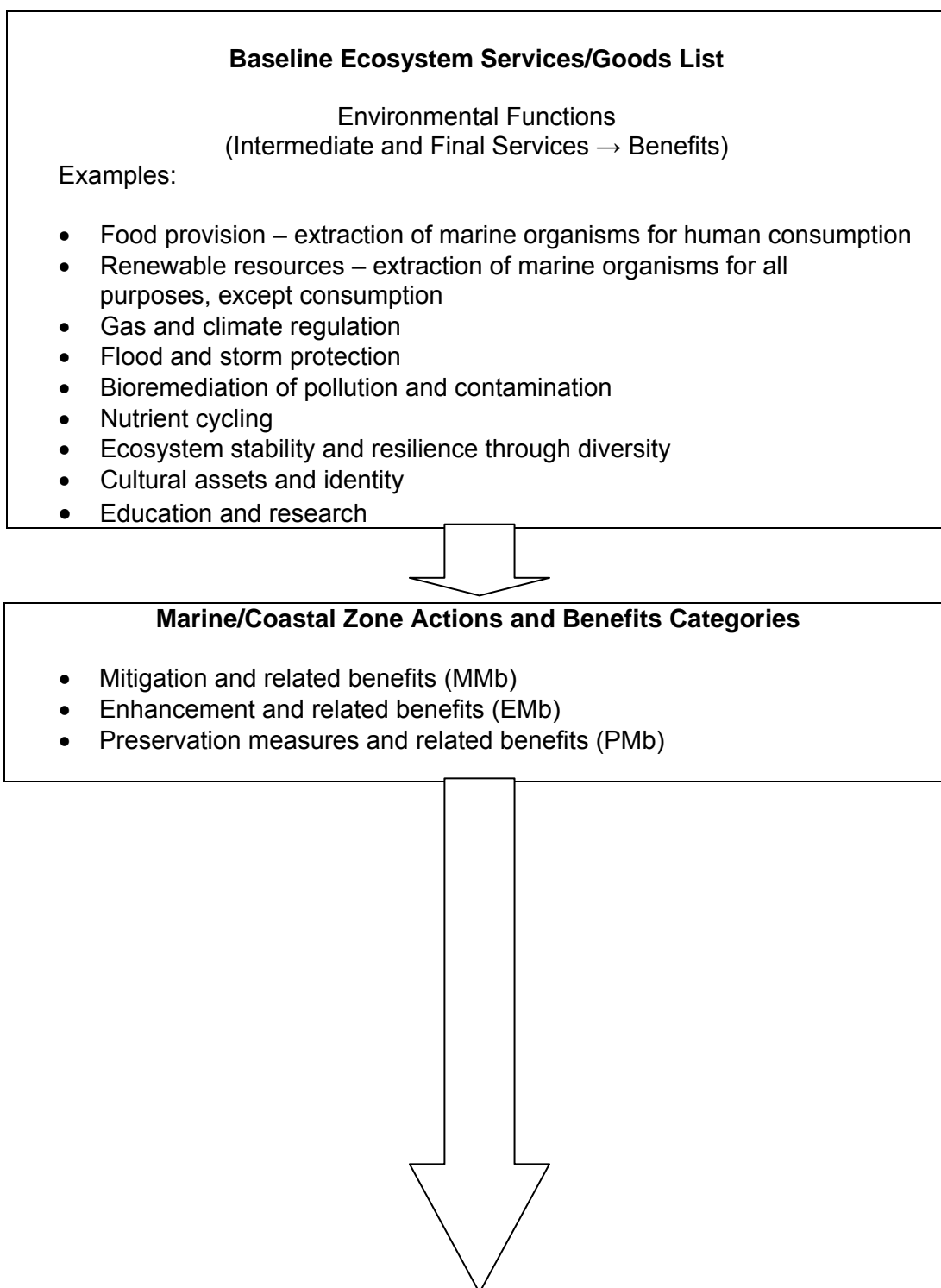
- **Mitigation of pollution and resource overexploitation problems** – the benefits (use values) that need to be valued are related to damage reduction and/or restoration measures, e.g. reduced flooding damage or sedimentation in navigation channels or restoration of wetlands, water treatment investment, changing farming practices in the catchments, etc, etc;
- **Enhancement of marine/coastal zone ecosystem goods and services** – actions (use value benefits), e.g. adaptation to change (autonomous adaptation, resilience), which increases the output of some product of service such as creation of artificial reefs to provide erosion protection or fisheries habitat etc. or the reduction of conflicts among or between various users of coastal ecosystems via pricing schemes or zoning, etc;
- **Preservation of unique marine/coastal ecosystems** – the benefits stem from setting aside and managing particular areas in order to preserve the natural ecosystem and two types of benefits can be involved. 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.

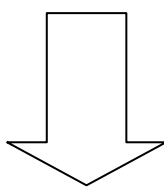
The marine and coastal zone interventions and their benefits (use and non-use values) can be linked to four environmental impacts/effects categories (relevant for human welfare):

- Direct and indirect productivity effects;
- Human health effects;
- Amenity effects (congestion); and

- Existence effects such as loss of marine biodiversity and/or cultural assets.

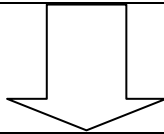
Different economic valuation techniques will be appropriate for each of the four broad effects categories, but it will not be possible to place meaningful monetary values on all the benefits (and some of the costs) of outputs from the marine/coastal zone . In particular the symbolic and cultural values assigned to some marine/coastal features and land/seascapes lie outside the monetary calculus and are conditioned by social preferences and norms arrived at, over time, through various forms of information transmission, art, literature film (see Annex 2 for more details). For any given policy issue, the following analytical sequence will prove useful in terms of scoping out an economic assessment:





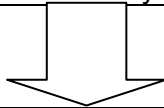
Use values and non-use values

- Direct and indirect use values including option value
- Non-use values, relating to existence and/or bequest motivations
- MMbs relate to direct and indirect use values plus option value
- EMbs relate to direct and indirect use values plus option value
- PMbs relate to direct and indirect use values, option values and existence value



Type of Effects Categories and Ecosystem Services

- Use values relate to productivity, human health and amenity effects
- Non-use values relate to existence and bequest
- Productivity effects directly related to, for example:
 - fisheries, agriculture, recreation/tourism, water resources, industrial production, navigation and indirectly to ecosystem processes yielding storm protection, flood alleviation, erosion reduction, sedimentation and waste assimilation, nutrient cycling etc
- Human health effects related to habitats, landscapes and cultural assets
- Existence value effects related to ecosystems and cultural assets.



Types of Economic Valuation Techniques	
Effects	Valuation Approach
Productivity	market orientated benefit valuation, using market prices of goods and services, based on changes in the value of output, or loss of earnings <ul style="list-style-type: none"> • For example, loss of fisheries output due to pollution, or recreational benefit loss through increase illness caused by polluted coastal waters
	surrogate markets benefits valuation including marketed goods, property values (hedonic pricing) and other land values, travel costs of recreation, wage differentials, compensation payments, damage costs avoided.
	cost terms in a cost-effectiveness analysis by using actual market prices of environmental protection inputs; known as preventative expenditures, replacement costs, shadow projects; defensive expenditure
Health	cost of illness measures, preventative and defensive expenditures, or survey-based valuation (see below)

Amenity	travel costs of recreation, properties/land values, or survey oriented methods such as contingent valuation, contingent ranking and choice experiments, using questionnaires to elicit individual willingness to pay or to be compensated valuations
Existence/ Bequest	only derived from survey-based methods

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 and services 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 (multi-criteria assessment) – see Figure 1.

The core of the economic assessment process therefore is to determine how changes in ecosystem services provision are translated into changes in welfare benefits (which can be plus or minus, 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.

Whatever methodology is used to conduct the assessment, all results should be subjected to a rigorous uncertainty/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 final services 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.

Most methods of economic assessment are concerned with determining the efficiency of policy options where efficiency is defined in a very narrow 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. 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 may be encompassed by a wider multi-criteria assessment.

Annexes 1 and 2 provide more details on the CBA/CEA methodology and the available monetary valuation techniques.

4.2 Some Key concepts

4.2.1 Weak and strong sustainability

Much of the economic debate about the definition of sustainable development takes as its starting point a very general definition of a sustainable state. This definition implies that the current generation must pass on a stock of capital to the next generation that is no less than it is now. This debate has developed to represent two very different views about the conditions that are necessary to realise the sustainable state defined above; these have come to be known as *weak* and *strong* sustainability arguments. These arguments differ in terms of the extent to which they regard that different types of capital are able to be substituted for one another. If capital is defined in a very broad sense as any economically useful stock then it can be classified as consisting of:

- physical, man-made capital – the machinery, buildings, etc. used to produce goods and services;
- human capital – not raw labour power, but the skills, knowledge and experience that individuals possess;
- natural capital – the goods and services that are provided by ecosystems, ranging from stocks of water, biomass or fertile land to nutrient cycling and climate stabilisation services;
- Social capital – cultural and other social norms including levels of trust and accountability in social institutions.

An advocate of an extreme form of weak sustainability would maintain that sustainable development can be achieved simply by transferring an aggregate capital stock value to the next generation that is no less than the current level. This assumes that there is perfect substitutability between the different forms of capital. This would mean, for example, that the current generation can degrade the stock of natural capital as long as they compensate for this by a proportionate increase in the stock of physical capital.

The strong sustainability standpoint, on the other hand, does not accept that perfect substitution possibilities exist between different forms of capital. Some elements of natural capital - for example the life support services of ecosystems - cannot be substituted by physical or human capital. Ecological assets that are essential for human survival and human wellbeing and which are not substitutable are classified as being *critical natural capital* and so should be protected according to this view.

The two extremes of the argument are presented above, in reality there are a number of more pragmatic 'middle ground' possibilities. However, setting the argument out in this way is useful in helping to highlight the types of economic assessment that are implied under different 'strengths' of sustainability and for making obvious the tradeoffs between capital types that these assessment methods attempt to consider. For example, the decision rule by which options are ranked using CBA is based upon choosing those options which maximise economic welfare – where economic welfare is explicitly measured in monetary terms. This implies that the welfare benefits that individuals derive from use (or simply from knowledge of the continued existence of ecosystems) can be translated into a monetary value that can then be compared to the cost of that option. Embedded within this is an assumption that man-made capital can be substituted for natural capital and hence CBA must be aligned with a position somewhere on the spectrum towards weak sustainability. To contrast with this, a position of very strong sustainability would allow no trade-offs between man-made and natural capital and so CBA would not exist in a world where strong sustainability was the sole guiding principle.

Degradation and loss of ecosystems, and subsequent loss of their associated services, constitute a reduction in natural capital. Whether or not this implies an unsustainable path depends on the extent to which one believes that the ecosystem services provided by natural capital can be substituted for by other forms of capital. Whatever the case there is a great deal of uncertainty about both the consequences of ecosystem service degradation and loss, and the ability to generate substitutes. Given this uncertainty, and the potential for catastrophic change, many would argue for a precautionary approach, in which case current rates of biodiversity and other natural capital depletion are a source of serious concern for sustained maintenance of human welfare. Annex 1 contains a section which summarises the various approaches that have been suggested to adapt economic CBA to better cope with uncertainty.

Most recently, it has been argued that the whole natural capital/stock and flow approach to environmental management has serious limitations and can serve to obscure the need for more radical reforms of institutions and governance. Much depends on how pressing the global sustainability constraints really are, but institutional and governance issues are clearly key parameters that need to be addressed in any meaningful sustainability dialogue and so far progress at the national and international level in this dimension has been very limited (Norgaard (2009) and see last section of annex 1 in this report).

Ideally ecosystems would be managed with sustainable development in mind. In practice, there are a number of acknowledged reasons why ecosystem degradation continues unabated. These reasons include both market failure and poor governance. One of the key causes of market failure is lack of information, and so the provision of information on the economic value of ecosystems can only contribute to (while not a guarantee for) better decision-making. This current lack of knowledge relates both to ecosystem functions and economic values. Poor knowledge of the mechanisms by which 'healthy'

ecosystems are maintained and are better able to withstand stress and shock (resilience) is a barrier to the development of effective management and assessment protocols.

4.2.2 Welfare, Benefits and Costs

Much of the text that follows refers to the effects of changes in ecosystem final services and goods on human society in terms of increases or decreases in benefits, costs, welfare, utility or human well-being. These terms require some definition. When we refer to benefits of a policy or project we mean that there has been (or, will be) some increase in human well-being or welfare associated with implementing that policy or project. Economists measure this increase in human well-being or welfare using the concept of utility. Utility is a measure of satisfaction: the more utility we have the more satisfied we are, or, alternatively the greater is our welfare or well-being.

Costs are the opposite of benefits. If the overall effects of a policy or project represent a cost to society this would mean that implementing that policy or project would result in a decrease in society's welfare or well-being and hence in the overall utility that society enjoys.

The problem with the concept of utility is that it is not directly measurable – so, how then do we compare situations where utility has been changed as the result of the implementation of some project or policy? Consider a simple example where we have one individual who enjoys a particular level of utility – we will call this U_0 – that is attained with an income of Y_0 , and which is associated with a given level of environmental quality – E_0 . Suppose then that the implementation of a new policy or project causes an improvement in the environmental quality that the individual experiences from E_0 to E_1 and that this improvement increases their utility from U_0 to U_1 : so they move from a state $U_0(Y_0, E_0)$ to $U_1(Y_0, E_1)$. As we have said we cannot directly measure this increase in utility, but we can indirectly by considering how much income this individual would be willing to give up in order to bring about this change. Hypothetically, the individual is considering two combinations of income and environmental quality that both give her/him the same level of utility, i.e. U_0 . In the first combination income is reduced and environmental quality is increased, and in the second, income is not reduced and environmental quality is not increased. The reduction in income that is required to make these two combinations equal represents what the individual is *willing to pay* for the change in environmental quality, i.e.:

$$U_0(Y_0 - WTP, E_1) = U_0(Y_0, E_0)$$

Alternatively an individual could be asked to consider how much additional income they would be *willing to accept* in order to give up the improvement in environmental quality, but still remain at the increased utility level U_1 , i.e.:

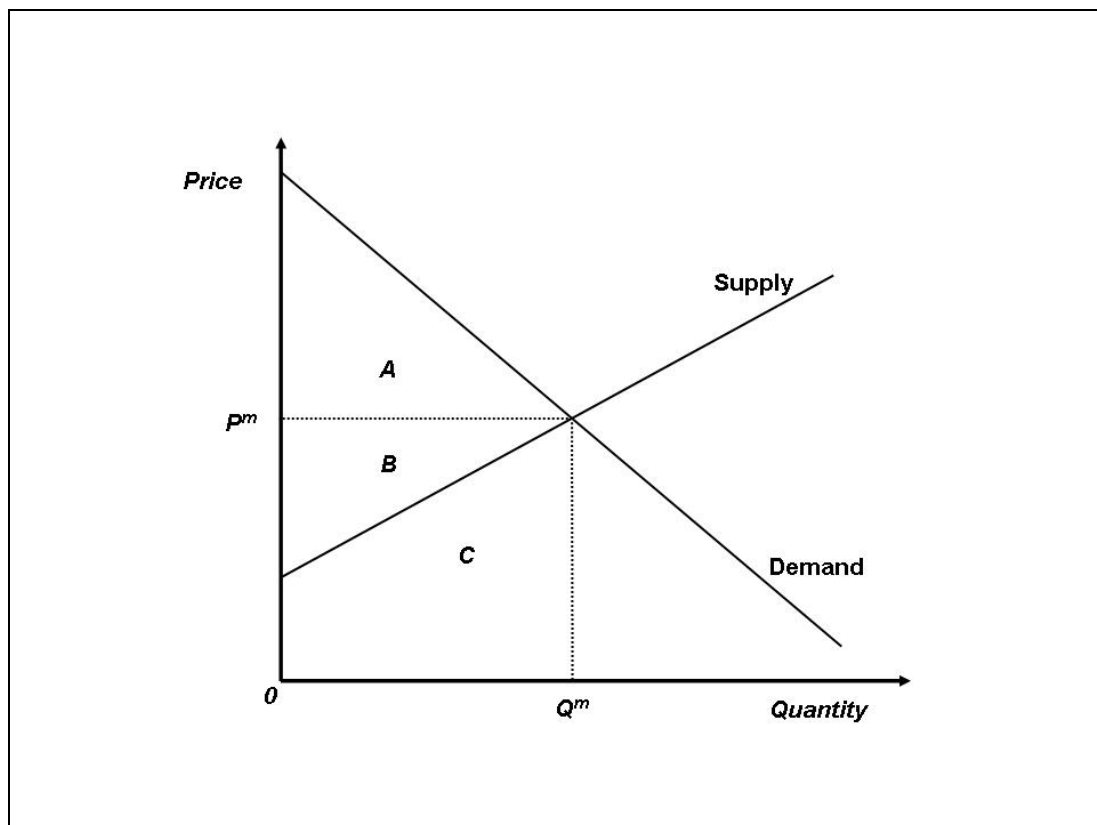
$$U_1(Y_0 + WTA, E_0) = U_1(Y_0, E_1)$$

Similar measures of change in utility can be developed for policy or project effects that cause deteriorations in environmental quality.

The basic principle that is at work here is that utility (or alternatively, welfare or well-being) can be indirectly measured in terms of the income that people are willing to give up in order to achieve some improvement; or, what they are willing to accept in compensation for foregoing some improvement. Willingness to pay (WTP) and willingness to accept (WTA) represent the monetary equivalents of changes in utility. Ways in which WTP and WTA can be estimated for goods and services which are not traded in markets will be detailed later in the report.

WTP equates to economic conceptions of value and it is useful to discuss this by reference to the demand and supply curves for a hypothetical good or service. To simplify things Figure 5 represents these curves as straight lines.

Figure 6 Willingness to pay, price and consumer surplus



The slope of the demand curve shows how much consumers are willing to pay for each extra unit of the good or service (i.e. it describes the marginal benefit they derive from each extra unit), and the demand curve slopes downwards because the benefit (utility) they derive from each additional unit declines with increasing quantity (known within economics as the law of diminishing marginal utility). The supply curve slopes upwards as the curve is derived

from the costs of production, as more is produced more inputs are required and this increases the costs of each additional unit produced (i.e. the supply curve is directly analogous to the marginal costs of the firm). Hence producers will only supply extra units for a corresponding increase in price.

The area under the supply and demand curves indicates the aggregate supply and demand respectively for the good or service (it is aggregate in the sense that it represents the sum of all the individual demands of all the consumers in this market, and the sum of supply from all the firms in this market). In a competitive, freely functioning market, a quantity Q^m of the good or service is traded at the market price P^m , which is the price at which demand matches supply. If quantities less than Q^m are traded, consumers are willing to pay more than the market price (the demand curve is higher than the level P^m), suggesting that market price alone is only a minimum estimate of the economic value or benefit derived. The area between the market price and the demand curve (triangle A) is the consumer surplus, or the additional utility gained by consumers above the price paid. Therefore, gross social benefits are the expenditure (areas B + C, or price multiplied by quantity) plus the consumer surplus (area A). The total cost of producing quantity Q^m is the area below the supply curve (area C). The area above the supply curve and below the market price is the producer surplus; this occurs because producers are willing to sell for less than the market price if the quantity traded is less than Q^m (the supply curve is less than P^m). The net social benefit is the consumer surplus (area A) plus the producer surplus (area B).

The point of this exposition is to make it clear that the price of a good or service and its economic value are distinct and can differ greatly: so, for example, water used for irrigation could have a very high value, but a very low price or no price at all (Turner et al., 2005).

A further point that should be made here is that demand curves and hence WTP are directly related to ability to pay. This has important implications relating to the distribution of gains and losses when economic analysis is undertaken that uses WTP principles. For example, CBA applies an equal weighting of gains and losses across all individuals and assumes that the prevailing distribution of income is socially acceptable. However, CBA can be modified by using equity and/or distributional weights which are determined using social or political criteria if there are particular distributional issues that need to be accounted for (Turner, 2007).

The type of socio-economic assessment that is advocated in this report seeks to incorporate the ecosystem services approach to policy and management into an economic assessment. So we now take a closer look at the ecosystem services and their economic valuation.

4.3 From Ecosystem functions to ecosystem goods and services

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 characterise a particular ecosystem; they are its functions, i.e. what the ecosystem does. Ecosystem functions can be grouped into five broad categories as follows:

- **Purification and Detoxification:** filtration, purification and detoxification of air, water and soils;
- **Cycling Processes:** nutrient cycling, nitrogen fixation, carbon sequestration, soil formation;
- **Regulation and Stabilisation:** pest and disease control, climate regulation, mitigation of storms and floods, erosion control, regulation of rainfall and water supply;
- **Habitat Provision:** refuge for animals and plants, storehouse for genetic material;
- **Information/Life-fulfilling:** aesthetic, recreational, cultural and spiritual role, education and research.

Many of these ecosystem functions inevitably lead to goods (benefits) that are consumed by humans, or which are essential for human survival (MEA, 2005). Ecosystem services and goods are defined in the next section and it is changes in these that we are interested in measuring and incorporating into economic analysis.

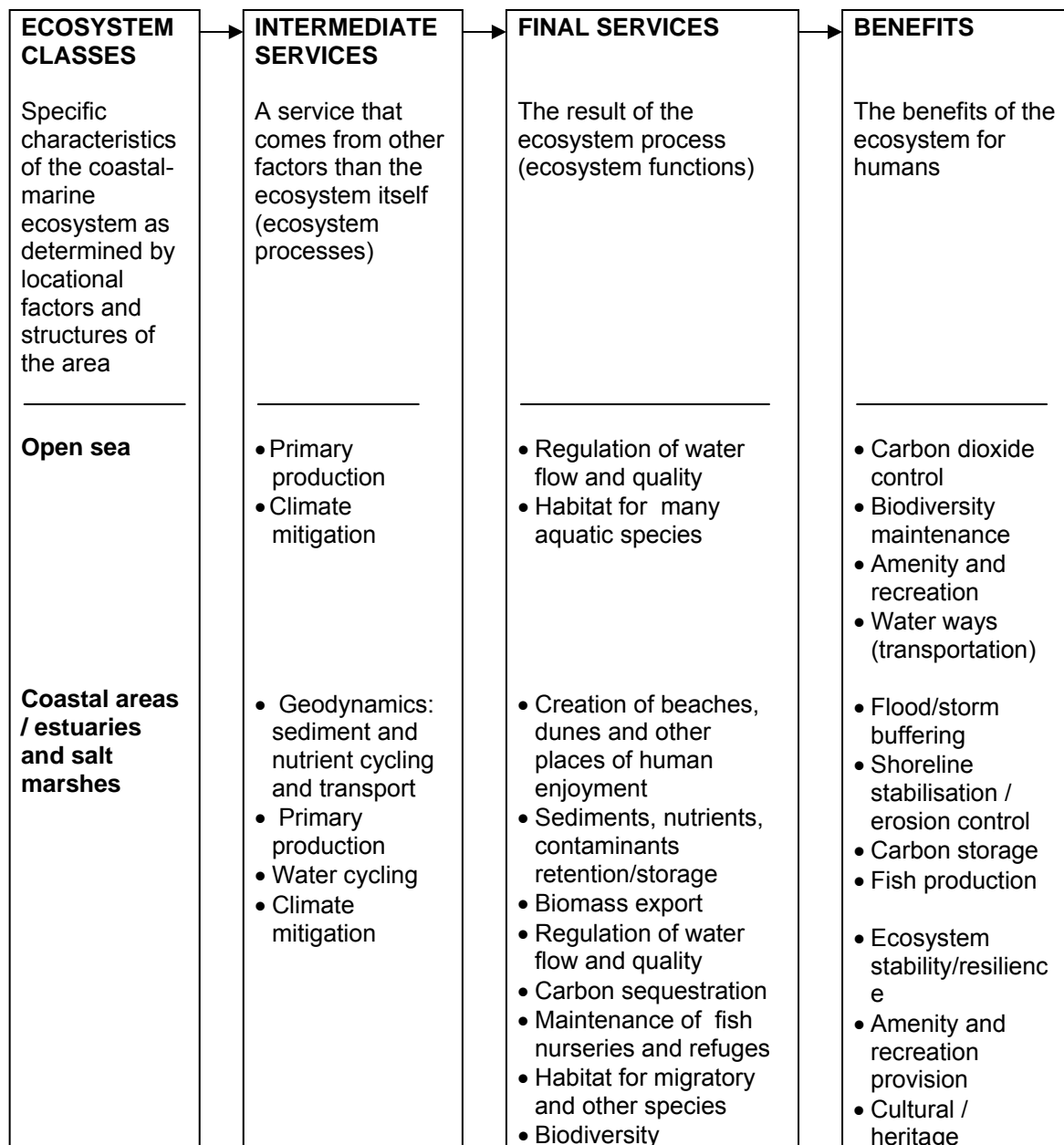
4.3.1 Ecosystem goods and services

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 coastal and related marine areas is maintained through a diversity of ecosystems – coral reefs, mangroves, salt marshes and other wetlands, sea grasses and sea weed beds, beaches and sand dunes, estuaries and lagoons, forests and grasslands. 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 7) (Crossland et al., 2005).

Coastal zone ecosystems are impacted by dynamic environmental change that occurs both ways across the land-ocean boundary. 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 inundation due to sea level rise. All coastal zone natural capital assets have suffered significant loss over the last three decades (e.g. 50% of 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; Maler et al., 2008; Barbier et al., 2008).

Figure 7 Classification of Coastal and Marine Ecosystem Services

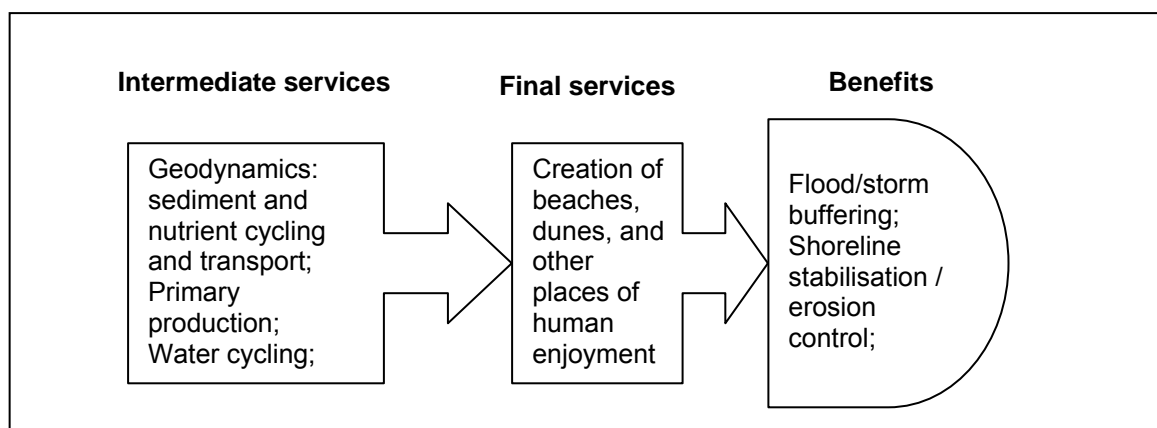


4.3.2 A conceptual framework for ecosystem services

Many definitions and classification schemes for ecosystem services exist (Daily, 1997; Costanza et al., 1997; Boyd and 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 an excellent 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 Maler et al. (2008) take national income accounting as their policy context. For economic 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.* see ecosystem services as being 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 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 in intermediate and final services, with the latter yielding welfare benefits (Figure 8).

Figure 8 Example of relationships among representative intermediate services, final services and benefits

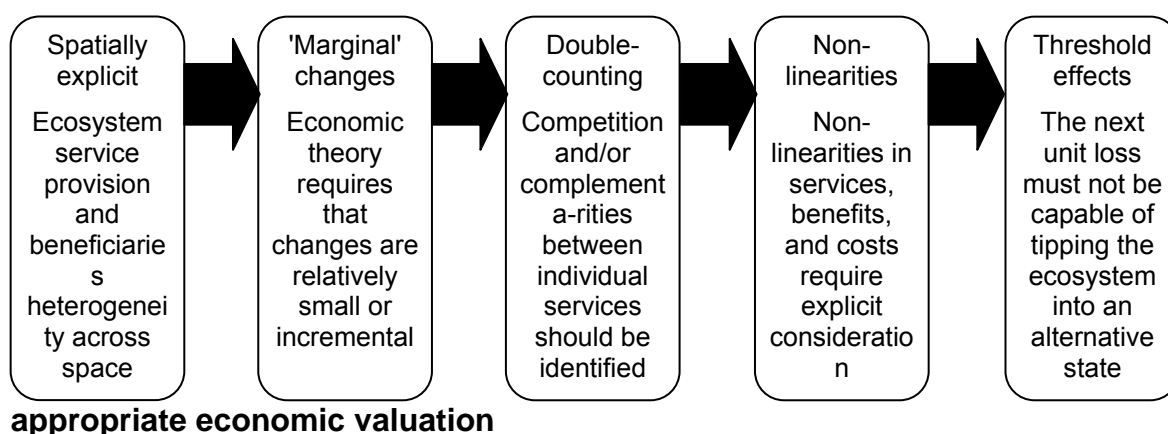


Adapted from Fisher and Turner (2008).

An intermediate service is one which influences human wellbeing indirectly, whereas a final service contributes directly. Classification is context dependent, for example, clean water provision 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.

In the economic literature, a number of issues can be identified as critical to the *appropriate* economic valuation of ecosystem services. These are: spatial explicitness, marginality, the double-counting trap, non-linearities in benefits, and threshold effects (see Figure 9).

Figure 9 Ecosystem Services Sequential Steps: A framework for



Source: Morse-Jones et al. (2008)

Spatial explicitness

It is critically important to first and foremost clarify the level of understanding (or ignorance) of underlying biophysical structure and processes through spatially-explicit models of any given ecosystem service. This contextual analysis must then encompass appropriate socioeconomic, political and cultural parameters in order to properly identify ecosystem services supply and demand side beneficiaries. The requirement for spatially explicit ecosystem valuation is based on recognition that ecosystem services are context dependent in terms of their provision and their associated benefits and costs. The importance of this point can be illustrated in the example of coastal wetland services provision (Andrews et al., 2006; Shephard et al., 2007; and Turner et al., 2007). One of the services provided by wetlands is carbon

storage but the net effect of this service is conditioned by the simultaneous release of methane. It turns out that the spatial location of the wetland and in particular the salinity condition plays a significant role in the carbon storage to methane emission ratio and the consequent global warming effect.

An essential component of the valuation approach that has rapidly emerged is the use of GIS techniques. Explicitly incorporating the spatial context is critical in obtaining unbiased estimates of both the costs and benefits of ecosystem provision, and, crucially, in enabling planners to identify the most economically efficient trade-offs. It is anticipated that the incorporation of spatial factors in ecosystem valuation is likely to become easier and more commonplace as access to GIS software and expertise increases (Bateman et al., 2006).

Marginality

Economics requires that for the valuation of ecosystem services to be meaningful such analysis should be conducted “at the margin”. This means focusing on relatively small, incremental changes rather than large state changing impacts. Given the scientific uncertainties which shroud ecosystem functioning, it is often difficult to discern whether a given change is ‘marginal’ or not and when thresholds are being approached or crossed.

Knowledge of the drivers and pressures on the ecosystems under study, as well as understanding of how the system is changing or might change from its current state is crucial. This has been called the system’s transition path (Turner et al., 2003; Fisher et al., 2009). It is important to know if the transition path is “stepped” as in the loss of a full coral reef system or shallow lake, or it is “relatively smooth” such as in species invasion into an area. By identifying the transition path, we can force the analysis to consider losses or gains in service provision or economic value between two distinct states of the systems.

While it is appropriate to consider, as far as is feasible, economic value in terms of marginal changes, a review of the existing empirical literature suggests that in fact very few studies do so. Maler et al (2008) explicitly undertake marginal analysis in estimating the accounting price for the habitat service provided by a mangrove ecosystem to a shrimp population. Their model evaluates changes to fisherman wellbeing for a *10 hectare* change in the stock of a mangrove forest of *4000 hectares* in size, obtaining an accounting price of \$200/hectare. In most cases, the ecosystem valuation literature has focused on valuing the stock, or the actual service flow. In some cases these analyses have been placed in a context of ‘change’ by drawing comparisons with alternative land use options.

Double counting

Another widely recognised issue concerns the potential problem of double-counting. This may occur where, competing ecosystem services are valued separately and the values aggregated; or, where an intermediate service is

first valued separately, but also subsequently through its contribution to a final service benefit. The value of a marine ecosystem for industrial fishing, for example, should not be added to the value of the same marine area for recreational fishing, since the former will likely preclude the later. Farber et al (2006) similarly note the problem of including aesthetic services and nutrient regulation in a case study of Plum Island coastal ecosystem. In essence, double-counting is a feature of the complexity of ecosystem services and the difficulty in understanding their multiple interactions.

Unfortunately, there are numerous cases where researchers have incorrectly summed values in order to obtain aggregate estimates of ecosystem value (evidence from Fisher et al., 2009). It is thus essential that the analyst has a clear understanding of the various overlaps and feedbacks between services when undertaking aggregation. Hein et al. (2006) suggest only including regulation services in valuations if '(i) they have an impact outside the ecosystem to be valued; and/or (ii) if they provide a direct benefit to people living in the area (i.e., not through sustaining or improving another service)' (p.214). Alternatively, the classification scheme recommended by Fisher and Turner (2008) as shown in Figure 2 helps to avoid the problem by drawing a clear distinction between intermediate services, final services, and benefits, the latter being the focus of economic valuation (see also Maler et al., 2008).

Non-linearities

The existence of non-linearities in ecosystem services provision adds further complexity to their valuation and subsequent management. Because many ecosystems typically respond non-linearly to disturbances, their supply may seem to be relatively unaffected by increasing perturbation, until they suddenly reach a point at which a dramatic system changing response occurs, for example, in the ecology of phosphorus-limited shallow lakes which can flip suddenly from one state to another. Further, in situations where non-linearities occur, one cannot make the assumption that marginal benefit values are equally distributed. For example, the storm protection benefit of a unit increase in mangrove habitat area may not be assumed to be constant for mangroves of all sizes due to non-linearities in wave attenuation (Barbier et al., 2008). If a cost-benefit appraisal assumes linearity, but service provision is in fact non-linear, policy option outcomes may be unnecessarily polarised. Correspondingly, for ecosystem valuation to better inform policy decisions, non-linearities need to be clearly understood and reflected in both ecological and economic analysis.

Barbier et al. (2008) have stressed that for some ecosystems (such as: coastal mangroves, salt marshes and other marine ecosystems) the services provided change in a non-linear way as habitat variables such as size of area alter. They claim that recognising such non-linearities opens up the choice set available to policymakers. In the case of mangroves and the storm buffering service they provide, it is argued that the non-linear supply of the buffering service (i.e. reducing as successive landward zones of the mangrove forest are crossed) means that some mangrove conversion (e.g. to provide space

for shrimp ponds) can be economically justified in cost-benefit terms. The authors note that an 'up to 20%' conversion rule seems to be an emerging policy principle. But such generalisations are dangerous because ecosystem services must be assessed in a spatially explicit manner and with due regard for uncertainties surrounding possible threshold effects. In the mangrove example it matters crucially where the shrimp ponds are located and what the current degradation status of the mangrove forest is. If the shrimp ponds are located on the seaward edge of the mangroves they will be prone to storm damage and lost productivity. If the mangrove has already experienced significant degradation it may be at or close to a threshold tipping point. Finally, mangroves (and other ecosystems) supply a range of interconnected services the value of which needs to be included in any economic benefit and loss account.

Threshold effects

A threshold effect refers to the point at which an ecosystem may change abruptly into an alternative steady state. For marginal analysis to hold true, the 'next unit' to be valued should not be capable of tipping the system over a functional threshold or 'safe minimum standard' (SMS). In practise, this requires knowledge of the amount of the SMS and its possible tipping threshold. Of course, due to the considerable uncertainty surrounding ecosystem functioning this introduces complexity since it is often far from clear when a threshold may be reached. For this reason, threshold effects pose especially complex policy and analysis challenges. Identifying this hazardous zone, in fact, will require expert input from ecologists, risk analysts and others, and may ultimately require ethical/political choices to be made and deliberatively agreed.

The challenge in incorporating threshold effects in ecosystem services valuation lies in our relatively limited knowledge of ecosystem complexity and interrelationships. Moreover, individual valuation studies frequently do not have the resources to undertake complex biophysical modelling. Consequently, the importance of threshold effects is often acknowledged in the valuation literature but rarely explicitly incorporated. Soderqvist et al (2005) apply the travel cost method to value the benefit of a bigger fish catch to recreational fishers in the Stockholm Archipelago. The results indicate that doubling the average spring catch per hour of Perch from 0.8kg to 1.6 kg amounts to a WTP of 56 SEK per angler. While on the surface this appears to be a small change, appropriate for marginal analysis, it is possible that the cumulative effect of doubling fish catch per hour could result in flipping the recreational fishery into an alternative state.

In summary, to be most useful for policy, services must be assessed within their appropriate spatial 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 SMS (MEA, 2005; Turner et al., 2003).

4.4 Total Economic Value

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 and services 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. A minimum configuration of ecosystem structure and process is required before final services and goods can be provided. Because there is uncertainty over what is or is not a sustainable ‘healthy’ functioning ecosystem state in many contexts a precautionary approach to management has much to recommend it.

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. Or cultural/symbolic values which groups of people have assigned to landscapes etc. However, apart from the problems of making the notion of intrinsic value operational, it can be argued that some people’s willingness to pay 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.

Marine/Coastal ecosystems provide a wide range of final services and goods of significant value to society - fisheries, transport medium, storm and pollution buffering functions, flood alleviation, recreation and aesthetic services, and so forth. In valuing such assets, it is important to capture the values to society of these characteristic services and goods. The use of the total economic value (TEV) classification enables the values to be usefully broken down into the categories shown in Figure 8. The initial distinction is between *use value* and *non-use value*. Use value involves some interaction with the resource, either directly or indirectly:

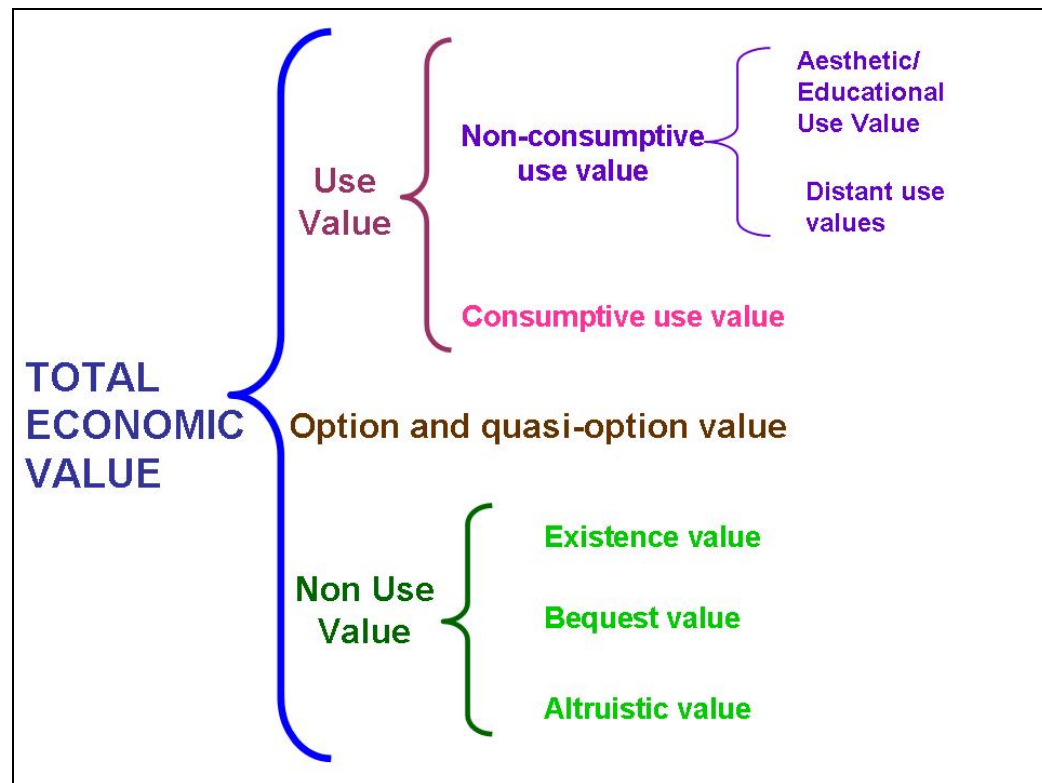
- *Direct use value*: involves direct interaction with the ecosystem itself rather than via the services it provides. It may be consumptive use, such as fisheries or timber, or it may be non-consumptive, as with some recreational and educational activities. There is also the possibility of deriving value 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.
- *Indirect 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.

- *Existence 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/intrinsic value divide.
- *Bequest 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.
- *Altruistic value*: associated with the satisfaction from ensuring resources are available to contemporaries of the current generation.

Figure 10 Total Economic Value



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

- *Option 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.
- *Quasi-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. The main problem when including the full range of ecosystem goods and services in economic choices is that many of these services are not valued in markets. There is a gap between market valuation and the economic value of many

ecosystem functions. To fill these gaps, the non-marketed services must first be identified and then where possible monetised.

TEV is derived from the preferences of individuals. When goods and services are exchanged in actual markets, individuals express their preferences via their purchasing behaviour. In other words, the price they pay in the market reflects how much, at the very least, they are willing to pay for the benefits they derive from consuming that good or service. For environmental resources which are not traded in actual markets, such behavioural and market price data are missing. Hence these resources generate non-market or external benefits. In addition to interpreting the market data, the methods of economic valuation provide several tools that may be employed to value benefits that are derived from non-market goods and services.

Choices between different policy options usually involve *marginal* changes in the provision of ecosystem goods and services. It is the marginal value of ecosystem services, i.e. the value yielded by an additional unit of the service, all else held constant, that will determine the consequence of trade-offs, i.e. the costs of losing or the benefits of preserving a given amount or quality of a service (Daily, 1997). In other words, the methodologies for estimating economic value relate to relatively small changes in ecosystem services, not to the totality of the functions themselves. Clearly the value of the latter is infinite, as without this stock of natural capital, there would be no life on earth.

4.5 Policy issues

Identification of a relevant policy issue is a key stage of the assessment process. The framework of an appropriate policy issue is necessary in order to:

- enable identification of suitable policy instruments, and;
- construct scenarios of possible future outcomes.

Which are, in themselves, steps that necessarily need to be taken in order to frame the context of the eventual analysis.

Typical **policy issues** within the regional seas/coastal zone include:¹

Table 2 Coastal Policy Issues

ENVIRONMENTAL ISSUE	EXAMPLES OF LOSSES IN ECOLOGICAL SERVICES AND ENVIRONMENTAL DEGRADATIONS
1. Increase in human population size and activities on coasts	
1.1. Artificial surfaces increase	- Natural habitat destruction : e.g. wetlands destruction with loss of flood control and pollutant abatement
1.1.1. residential and commercial facilities that have continued to	

¹ Please note that this is not an comprehensive list.

be constructed in risk prone zones (flooding and inundation)	ecological functions - arable land destruction with loss of the fertile soil particles for cropping
1.2. Increase in human activities intensity	- Loss in fish stocks provided by rivers and coastal waters due to the decrease in water oxygen concentration suffocating aquatic species - Heavy metals responsible for cetaceans beaching
1.2.1. Increasing eutrophication due to more, and more intensive, agriculture, industries and domestic pollution in nitrogen and phosphates 1.2.2. Direct discharge of waste by industries into rivers or into the sea	
1.3. Recreational users' conflicts	
2. Increase in shipping activities intensity	
2.1. Increase in pollutant discharge to the sea	- Affect sea biodiversity and coastal water quality with consequences on the fishery and tourism sectors
2.1.1. Oil spills and chemicals by shipwreck 2.1.2. Tributyltin (TBT) antifouling paints	
2.2. Increase in NOX emissions by boats (precursor of greenhouse gas, ozone, responsible for sea level rise)	- More frequent and violent coastal floods responsible for destruction of habitats and arable soils, soil salinization, coastline erosion, etc.
3. Increased intensity in maritime and river activities	
3.1. Construction of river dams and channelization of rivers	- Erosion caused by deficit in sediment causing coastline retreat - Losses in natural barriers to floods (dunes, wetlands...)
3.2. Destructions of natural habitats (dune barriers, wetlands...)	
3.3. Offshore sand and gravel mining affecting fisheries and habitats	
3.4. Use of sand for construction	
3.5. Aquaculture development	- Decrease in biodiversity with impact on the fishery sector
3.5.1. Overfishing 3.5.2. Eutrophication	
3.6. Wind farm development	
3.6.1. Bird deaths 3.6.2. Silting up (formation of sand banks modifying the ecosystem)	
3.7. Coastal armouring	
3.8. Introduction of exotic species (through water ballast and fouling)	
4. Fossil fuel consumption trends	

4.1. Global warming (due to GHG emissions)	
4.1.1. Sea temperature increase	<ul style="list-style-type: none"> - Losses and redistribution of marine species - Enhanced conditions for growth of algal blooms toxic to fish and humans
<p>4.1.2. Increase in water use for irrigation</p> <p>4.1.3. Rising sea level and increase in the frequency and violence of coastal floods</p>	<ul style="list-style-type: none"> - Ground water exhaustion enhanced by the combination of higher temperatures, growing coastal population and tourism - Destruction of human infrastructure caused by flooding - Salt water intrusion from sea into aquifers (salinisation of arable land and drinking water) due to decrease of water table levels and floods

Annex 4 in this report summarises the financial and economic valuation studies that have been undertaken or are on-going in OSPAR and HELCOM member countries.

5 Conclusions

The DSS set out in earlier sections of this report takes a sequential approach. It can help to facilitate future cooperation between countries as called for at the fifth North Sea Conference in Bergen in 2002. It would also support efforts to develop a European Marine Strategy focused around sustainability principles. The DSS could inform OSPAR operational practice by giving guidance on which activities should be taken forward in the short term as priority actions. The DP-S-I-R framework could be used to identify OSPAR priority environmental change sequences (chains) and then to focus analytical efforts. The implications for EU Marine Policy could also be clarified at this stage. OSPAR could then move on to a futures scenarios exercise in which it agrees a BAU and relevant alternative future states of the world, in order to 'model' relative change in the Marine environment.

The detailed meaning of the term 'socio-economic' analysis could also be clarified and agreed. A lot of good work is being undertaken in member countries on a range of pollution contamination and ecological resource loss problems. However, the financial analysis which supports some of this work needs to be extended into a fuller economic cost-benefit (CBA) format. Further modifications of the economic CBA can then follow which would incorporate fairness and distribution equity concerns (i.e. who gains or loses from environmental change). This social analysis can include qualitative as well as quantitative assessments, highlighting unemployment, loss of cultural assets/identity etc problems (multi-criteria assessment). The intra-country work can be viewed as components of a wider spatial assessment at regional sea level. But ultimately an inter-country assessment exercise (including

valuation and distributional analysis etc) will be required. The cooperative working practices which would support such an exercise could be trialled and 'polished' during the initial scoping and scenario-building stages of the recommended DSS.

Annex 1: Methods for economic assessment

Figure 1 at the start of this report summarised the initial analytical step which is to decide on whether a full cost-benefit analysis seeking to determine net economic benefits from a project, policy or programme of coastal intervention, or a more constrained cost-effectiveness analysis is required. In the latter context, a range of options are usually assessed to see which yields the desired outcome, e.g. achievement of a given water quality status, at least cost to society.

This section provides some brief detail on the initial economic assessment steps that we would envisage taking place. The choice between CBA and CEA is determined by the nature of the policy problem under scrutiny. If the problem is one of meeting some environmental standard, complying with a law or achieving a target then finding the least cost way of achieving this by completing a CEA is the appropriate action. If the problem is one of choosing between an number of different possible policy or project options which do not involve compliance with standards or targets then CBA is the most appropriate assessment tool.

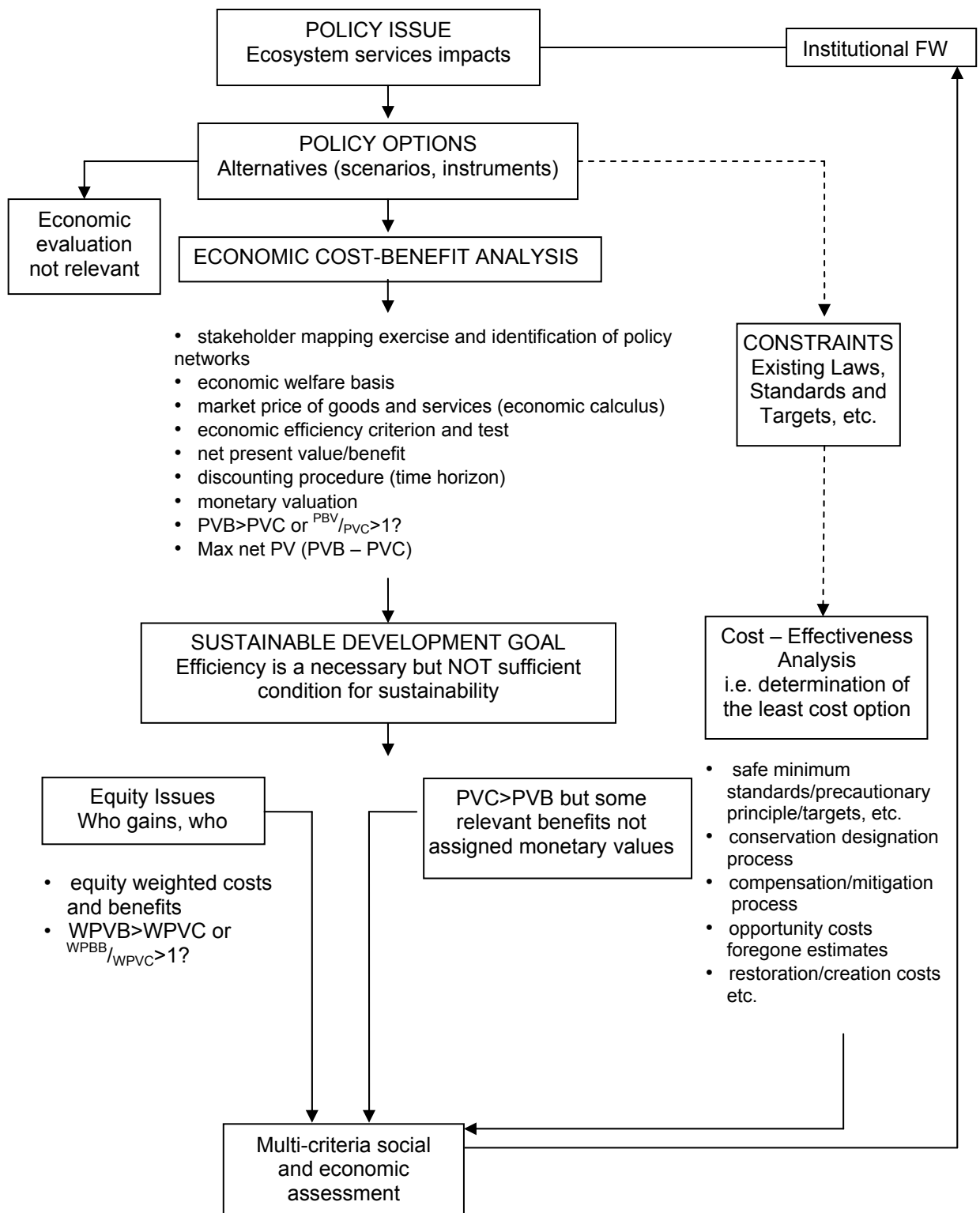
If the situation is one where monetary valuation is not possible or appropriate then CEA and CBA should be replaced with a multi-criteria assessment process.

A second stage of the assessment process may involve the use of regional economic methodologies such input-output analysis. Where the policy or project or change (eg loss of a fishery) is likely to generate a number of secondary regional effects – in terms of employment for example – then this stage may be contemplated in order to model these effects. There are however well known weakness with I-O models. The results of these analyses can then stand alone or can be fed back into the CBA.

Much more detail on CBA and CEA can be found in Pearce et al. (2006).

See Figure 1 below which is reproduced from first section of report.

Figure 11 Analytical steps for economic assessment methodologies



Cost benefit analysis

CBA is a means of project or policy appraisal. It involves identifying and measuring, in monetary terms, as many of the costs and benefits as possible that relate to a particular project or course of action. This helps to determine whether the project or policy will produce a net gain or loss in economic welfare for society as a whole. As a rule, a project is deemed to be efficient if total benefits exceed total costs. A simplified overview of CBA methodology is outlined in Box A1.1.

A CBA compares the costs and benefits of different policy options in monetary terms. The results of this analysis can be interpreted as a B-C ratio, i.e. total benefits divided by total costs, where a ratio larger than one indicates that the policy measure is economically beneficial, or as a NPV, that is the present value of the net benefits where a positive NPV indicates a welfare improvement. Strictly speaking, only those costs and benefits are included in a CBA that can be quantified in monetary terms. However, it will hardly ever be possible to monetise all impacts all the time: those impacts that cannot be monetised are often left out of the analysis. Non-monetised impacts, if considered relevant, can nonetheless be included in a qualitative discussion accompanying the discussion of the CBA results.

The theoretical foundations of CBA can be summarised as follows (Pearce et al., 2006):

- Benefits are defined as increases in human well-being (utility).
- Costs are defined as reductions in human well-being.
- To pass the economic efficiency test on cost-benefit grounds, a project, policy or course of actions' social benefits must exceed its social costs.
- "Society" is simply the sum of individuals.
- The geographical boundary for CBA is usually the nation but can be extended to wider limits, given appropriate data.
- Aggregating benefits across different social groups or nations can involve summing willingness to pay/accept (WTP, WTA) regardless of the circumstances of the beneficiaries or losers, or it can involve giving higher weights to disadvantaged or low income groups. One rationale for this is that marginal utilities of income will vary, eg higher for the low income group.
- Aggregating over time involves discounting. Discounted future benefits and costs are known as present values.
- Inflation can result in future benefits and costs appearing to be higher than is really the case. Inflation should be netted out to secure constant price estimates.
- The notions of WTP and WTA are firmly grounded in the theory of welfare economics and correspond to notions of compensating and equivalent variations.
- WTP and WTA should not, according to past theory, diverge very much. In practice they appear to diverge, often substantially, and with

WTA > WTP. Hence the choice of WTP or WTA may be of importance when conducting CBA.

Box A1.1: An outline of CBA methodology

The main stages of a CBA are as follows.

1. Definition of the details of each feasible project, policy or management option including the 'do nothing' option.
2. Determining the spatial and temporal scales of the analysis, i.e. over what population is it appropriate to sum the costs and benefits? and, over what time period do the costs and benefits arise?
3. Identification of the costs and benefits and their monetary values. Monetary value may be based on the market value of a good or service or on its replacement cost (if that can be calculated), or, in the case of some environmental goods and services, by use of various valuation techniques. To enable valid comparisons, all monetary values must refer to a common point in time – the base year – to give 'present' values. A standard 'discount rate' is applied so that costs and benefits of projects with varying time scales can be compared.
4. The economic efficiency of various options are assessed through comparing either their 'benefit-cost ratios', i.e. the present value of benefits divided by the present value of costs, or their 'net present values', i.e. the present value of benefits less the present value of costs.
5. A sensitivity analysis should be included within a CBA, to assess the impact on the benefit cost ratio and/or net present value of changes in the values of central parameters, e.g. the value of costs and benefits or the discount rate. By examining the impact that increasing costs (or reduced benefits) may have on the net present value, the break-even point can be determined whereby the scheme would be no longer justifiable.

There are numerous critiques of CBA. Perhaps some of the more important are:

- The extent to which CBA rests on robust theoretical foundations.
- The fact that the underlying "social welfare function" in CBA is one of an *arbitrarily* large number of such functions on which consensus is unlikely to be achieved.
- The extent to which one can make an ethical case for letting individuals' preferences be the (main) determining factor in guiding social decision rules (Turner, 2007).

CBA can provide a very useful and reliable input into the decision-making system, provided that it is carried out fully and impartially. However, translating all the costs and benefits of a project, policy or management scenario into monetary terms can be impractical or not meaningful. It should be remembered that CBA only provides an aid to decision making and that the most cost efficient option may not be the most appropriate on other grounds. In these situations multi-criteria analysis (MCA) can provide an alternative as it permits the inclusion of measurable non-monetary criteria into the assessment and explicitly allows for stakeholder deliberation and dialogue.

Finally, the whole history of neoclassical welfare economics has focused on the extent to which the notion of economic efficiency can or should be separated out from the issue of who gains and loses – the distributional incidence of costs and benefits. Various “schools of thought” have emerged. Some argue that distributional incidence has nothing to do with CBA: CBA should be confined to “maximising the cake” so there is more to share round according to some morally or politically determined rule of distributional allocation. Others argue that notions of equity and fairness are more engrained in the human psyche than notions of efficiency, so that distribution should be considered as a prior moral principle, with efficiency taking second place. Yet others would agree with the second school but would argue that precisely because efficiency is “downgraded” in social discourse that is all the more reason to elevate it to a higher level of importance in CBA. Put another way, one can always rely on the political process raising the equity issue, but not the efficiency issue. Certain minimum requirements for practice emerge. At the very least, a “proper” CBA should record not just the aggregate net gains from a policy, but the gains and losses of different groups of individuals.

Cost-effectiveness analysis

The purpose of a cost-effectiveness analysis (CEA) is to find out how predetermined targets, e.g. threshold values for nutrients or other pollutant loads in a catchment/coastal waters can be achieved at least cost. Theoretically speaking, the least cost allocation of pollution abatement strategies is found if the marginal costs of the proposed measures are equal. The marginal costs of these abatement measures can for example be defined as the increase in total abatement costs when pollution loads are decreased by 1 ton or 1 kilogram per year. As long as marginal costs are not equal, it is theoretically possible to obtain the same level of pollution reduction at lower costs by shifting emission reduction from high cost measures to lower cost measures.

The steps involved in conducting a CEA are described below:

Step 1: Define the environmental objective involved

Step 2: Determine the extent to which the environmental objective is met

Step 3: Identify sources of pollution, pressures and impacts now and in the future over the appropriate time horizon

Step 4: Identify measures to bridge the gap between the reference (baseline) and target situation

Step 5: Assess the effectiveness of these measures in reaching the environmental objective

Step 6: Assess the costs of these measures

Step 7: Rank measures in terms of increasing unit costs

Step 8: Assess the least cost way to reach the environmental objective

These steps are taken in sequence, but important feed-backs may exist between steps. As information becomes available about the problem, the source-effect pathway and possible solutions, the same step may be revisited several times. The outline of the various steps shows that carrying out a CEA is a multi-disciplinary exercise, requiring the input of and collaboration between different scientific disciplines, such as natural scientists, economists and technical engineers, but also the input of policy and decision-makers as they determine the scope and objective of the analysis.

A number of approaches are used in practice at varying levels of complexity, scale, comprehensiveness and completeness for carrying out a CEA. A distinction is made between bottom-up and top-down approaches. The bottom-up approach focuses on technological details of measures and their impact on individual enterprises (micro level), whereas top-down approaches usually consider the wider economic impacts of pollution abatement measures and strategies, often without detailed technical specification of the proposed measures (macro level). Bottom-up approaches can also be characterised as technical engineering approaches, often including detailed information about the technical characteristics of production processes and only limited information about the financial engineering costs of emission abatement technologies. Top-down approaches on the other hand focus more on the economic relationships and consequences involved and less on the technical specification of measures.

Multi Criteria Analysis

MCA is a framework which allows decision-makers to evaluate and rank a range of different management options according to a set of well-defined evaluation criteria. In its essence it is a very simple process to understand: the relevant criteria for decision-making are identified, an assessment is made of the impacts on these criteria of the considered management options, and these options are then ranked in terms of their impacts on the criteria. Additionally, where there may be a number of different users of a resource

who have differing preferences and priorities then MCA allows these to be incorporated in the form of weighted rankings of different options. Decision-makers can then compare unweighted rankings of management options with weighted rankings to gauge the level of support for, and possible impact of, their decisions. However, although the principles underlying MCA are simple the analysis itself is not: MCA is not an easy option since the process of weighting and ranking can involve some sophisticated statistical and programming techniques. Further, protocols are required to ensure that group-based decisions/outcomes are arrived at in as 'democratic' a way as is feasible and are not dominated by any particular special interest group/position.

The MCA framework

MCA should be viewed as a framework for analysis rather than as a straight alternative to appraisal methodologies such as cost-benefit analysis. In fact, MCA can integrate the results of CBA and other appraisal techniques to allow decision-makers to choose the most appropriate course of action. MCA also differs from CBA and related methods (which are much more unified in the techniques they involve) since the term covers a wide range of related, but differing techniques (examples include: multi-criteria decision analysis, multi-attribute utility theory, the analytic hierarchy process, and fuzzy set theory). Here we describe the MCA process in very simple terms. If required, readers can find further detail on MCA in the sources detailed in Box A1.2 below.

Box A1.2: Further information on MCA

David L. Olson (1996) *Decision Aids for Selection Problems*. Springer Series in Operations Research. Springer-Verlag, New York.

A clearly written book that outlines the process of various MCA approaches. Contains several illustrative case studies.

Janssen, R. (1994) *Multiobjective Decision Support for Environmental Management*. Kluwer Academic Publishers, Dordrecht.

An easily understandable text that supports the DEFINITE MCA computer program.

Keeney, R. L. and H. Raiffa (1993) *Decisions with Multiple Objectives: Preferences and Value Tradeoffs*. Cambridge University Press, Cambridge.

A very much more technical book than those listed above. Explains the theory underlying MCA techniques.

Department of the Environment, Transport and the Regions (2000) *Multi-criteria Analysis Manual*, DETR, London.

Available online at:

http://www.odpm.gov.uk/stellent/groups/odpm_about/documents/page/odpm_about_608524.hcsp

This is a manual produced to guide decision-making within the UK civil service that provides a clear and reasonably brief overview of various MCA techniques.

Dealing with Uncertainty, Irreversibility and Related Concepts

This note is intended to clarify the issues which surround the notion of **irreversibility** and related concepts of **threshold effects** and **tipping points**. While all these terms are now in use in the environmental conservation and economics literature, we find it helpful to refer to thresholds in the context of individual ecosystems or landscape ecology limited to the regional spatial scale. This reserves the term, tipping points, to describe global scale system and subsystem non linear and abrupt reactions to environmental change pressures (Rockstrom et al., 2009).

Ecosystems function via feedbacks between different components of structure and process. When the feedback effects are positive any given initial perturbation (stress or shock) of the system will be amplified, when a positive feedback occurs the prevailing state of the system may be such that a complete switch into a different state is triggered (a classic case is the enrichment of shallow lakes via excessive N & P inputs from the surrounding

catchment, causing abrupt change in water quality and aquatic plant and fish etc communities). The initial ecosystem state prior to the 'flip' is then a threshold or a bifurcation. The capacity of an ecosystem to 'absorb' stress or shock and remain in its prevailing state is known as resilience. It is still a matter of scientific debate whether greater diversity in ecosystems provides a buffering capacity (greater stability or resilience) and in which specific contexts (Worm et al., 2009). A further degree of uncertainty surrounds the question of whether the ecosystem state change is reversible or irreversible in the future. This is a far from straightforward question and we currently lack sufficient scientific and other knowledge to be able to offer robust prescriptions. In the shallow lake example cited earlier, it is the case that remedial management actions (such as sediment pumping, N & P abatement etc) can restore water quality & other losses. But even in this case, the timing and extent of the necessary abatement programme is not clear cut, with adverse consequences for the overall costs of action. In more complex contexts, irreversibility is even more difficult to pin down, either because of current technological / scientific data and means deficiencies, or impracticability constraints in the form of significant cost burdens and governance limitations. Future scientific and technological breakthroughs and socio-economic conditions and preferences may or may not lead to less constraints, but in any case they are currently only predictions or complete unknowns.

Given that information about ecosystem functioning and dynamics under contemporary environmental change conditions (local through to globalisation) is incomplete, there is a positive probability that a given ecosystem in a given change context will be pressurised into a thresholds zone and across a point, causing it to flip to a less desirable new state. The probability of flipping is lower as resilience is maintained / increased and management interventions to 'conserve' resilience are therefore important. Resilience capacity can be regarded as capital stock (natural capital) which yields an insurance service and benefit (Maler et al., 2008). As a stock, in principle it has an accounting price, defined as, the change in the expected change in net present value of the expected future ecosystem services resulting from a marginal change to resilience today (Maler et al., 2008). In practice, data (time series and other) constraints have so far precluded the monetary valuation of this insurance service, with the one exception of an agroecosystem study in Australia.

Two different approaches have been put forward in the environmental economics literature as coping strategies for the irreversibility problem. The first was a modified CBA method, known as the Krutilla-Fisher (K-F) approach. It laid down that in relevant preservation versus development situations, the benefit of the preservation option should be factored into the CBA equation. Preservation benefits forgone should be treated as part of the costs of development and should be assumed to increase through time because of the relative price effect. The development benefits should have an offsetting discount factor, in addition to the 'basic' discount rate because of 'technological obsolescence'. It is also the case that the present value of development can be very sensitive to the preservation relative price effect and the obsolescence factor (Pearce & Turner, 1990). Given the prevailing

information gaps, it is recommended that the benefit of the doubt be given to preservation over development in all cases where benefits and costs are reasonable closely balanced. The K-F strategy was designed to cope with ecosystem and landscape asset losses characterised by uniqueness and national / international significance. The dilemma was how to cope with possible irreversible losses of unique assets such as designated national park lands. But it may be the case that some potential ecosystem losses are of 'local' significance and 'uniqueness'. In these cases a conservation versus development trade-off needs to be addressed.

In these 'local irreversibility' situations a 'shadow project' approach in which sustainability considerations are integrated into the CBA calculus may be relevant. The decision maker is asked to consider a range of decisions about development options and impose a sustainability constraint into the decision support system and process (ie to keep the stock of natural capital (Kn) constant over time by suitable compensatory expenditures). The sum of the ecosystem damage done by a whole sequence of development projects would have to be offset by separate projects within the 'portfolio' of decisions being made. These compensatory projects would not have to pass the positive B-C ratio test (Barbier, Markandya & Pearce, 1990). The precise form of 'acceptable' compensation will vary from context to context and is an under-researched area. Roach & Wade (2006) for example have examined the use of so-called habitat equivalency analysis which estimates ecological service loss and then scales restorative ecological compensation to offset the damage impact.

A second line of argument if irreversibility concerns are relevant incorporates the notions of the precautionary principle and the safe minimum standard. There is a line of reasoning that can link ecosystem diversity and resilience maintenance (with 'primary' / 'glue' / 'infrastructure' values in nature, alongside the 'insurance' value noted earlier) together with a support for the precautionary principle and strong sustainability (Gren et al 1994, Turner et al., 2003). The precautionary principle is itself shrouded in ambiguity and CBA can provide a useful filter for it if a 'safe minimum standards' (SMS) interpretation covering species, habitats and ecosystems is accepted (Crowards, 1998). This goes back to the work of Ciriacy-Wantrup (1952) and Bishop (1978) in which it was advocated that a project should be rejected if irreversible losses of nature could consequently occur, unless the social costs of doing so were prohibitive (a modified minimax rule). Thus decision makers facing the prospect of very high preservation or conservation costs might choose to sanction a development option even though it carries a small risk of significant ecosystem damages. Nevertheless, judging what is or is not an "unacceptable large" or "tolerably low" social cost can be informed by ecology, economics, risk analysis etc, but ultimately is a 'political' call. Ethical and political choices will have to be made and deliberatively agreed. Over time, the aim should be to improve our understanding of ecosystem functioning so we can move towards more situations where we are dealing with risk rather than uncertainty.

It has recently been pointed out that, at the global spatial scale, many subsystems of Earth are sensitive to threshold effects and if a 'tipping point' is crossed unacceptable environmental change could be triggered (Rockstrom, 2009). The claim is that most of the thresholds can be defined by a critical value for one or more control variables. Nine processes have been identified (climate change; rate of biodiversity loss; interference with the nitrogen and phosphorus cycles; stratospheric ozone depletion; ocean acidification; global freshwater use; change in land use; chemical pollution and atmospheric aerosol loading) the first three of which have already reached the threshold zone.

Rockstrom et al adopt a precautionary safe minimum standards perspective and propose that the planet must be kept in a "safe operating space" through the observance of quantified boundaries. In the case of biodiversity loss they advocate a boundary of ten times the background rates of extinction. Because of the many gaps in our knowledge this boundary should be considered as preliminary, but it seems clear that the current rate of species loss (100 to 1,000 times more than what could be considered natural) will lead to significant reductions in ecosystem resilience. A further concern is that globalisation has resulted in a rate and extent of economic activity sufficient to pressurise a range of earth processes simultaneously. This means that the planetary tipping point boundaries are tightly coupled and piecemeal abatement strategies are unlikely to be sufficiently effective.

The implementation of this global safe minimum standards strategy will be controversial and will require concerted and targeted science and social science research efforts to underpin it. But most of all it requires a radical overhaul of the governance processes controlling international trade and finance and resource exploitation etc. (Norgaard 2009). In the interim, recent work by Lenton et al (2008) has proposed the use of early warning systems which identify systems that are likely to cross 'tipping points' and are relevant to policy and accessed by humans ("tipping elements"). Historical data and predictive modelling (e.g. degenerative fingerprinting) may then be used to locate tipping points.

Annex 2: Economic Valuation of Ecosystem Goods and Services

Introduction

The objective of economic valuation is to measure the strength and direction of the preferences of individuals' and hence the total economic value of a resource. This is achieved using the behavioural concepts of buyers' maximum WTP and sellers' minimum WTA for the good or service of concern. A variety of techniques can be used that provide information on values either by the analysis of data generated by actual market transactions, or using proxies to such data when markets for the goods and services of concern do not exist. In a market, individuals with a WTP lower than the market price will not purchase the good in question. Those with WTP equal to or higher than price will purchase the good. The excess of WTP over market price is known as 'consumer surplus'. This is the net benefit an individual receives from the consumption of a particular commodity. For a market good, total WTP is comprised of total consumer surplus (over all units of consumption) and total price paid, i.e. total expenditure. For environmental goods which are not traded in markets and hence do not have prices, WTP is wholly comprised of consumer surplus.

Environmental policy can be supported by non-market valuation since valuing changes in environmental goods and services in monetary terms makes it possible to directly compare the non-market impacts of a particular decision to market or financial benefits (revenues) and costs which are usually much easier to identify. Hence by estimating the value derived from intact ecosystems it is possible to present a case for conservation that is directly comparable to estimated returns from conversion of ecosystems or exploitation of ecosystem goods.

Several approaches can be used to estimate the value of ecosystem goods and services. These approaches fall into two groups:

- techniques that estimate economic values – valuation approaches, and;
- those that produce estimates that are equivalent to prices – pricing approaches.

This grouping arises because knowing the price of a given good informs us only of the cost of purchasing that good and not its value. As we have previously established, WTP, which is the appropriate measure for TEV, consists of **both** the price paid to purchase a particular good, as well as consumer surplus. Pricing approaches, or cost based measures are unable to capture the consumer surplus element of value and so must be regarded as only a partial measure. However, whilst valuation approaches may be theoretically correct, pricing approaches are often used to value various aspects of ecosystem value. This is because valuation approaches are often very expensive and time consuming to undertake and so price/cost based

techniques are common where time and resources are limited. In addition, pricing approaches can be useful in providing rough monetary estimates of environmental goods and services that might otherwise remain unvalued in the absence of other, more difficult to obtain (and often expensive), evidence. These approaches are presented below.

Valuation Approaches

Stated preference methods

Stated preference methods directly elicit individuals' preferences for non-market goods through the use of surveys based on simulated markets. In contrast to other valuation approaches, these methods can also estimate the non-use component of TEV (as well as other components). In the case of ecosystem goods and services non-use value may be significant, particularly for irreversible impacts.

The main forms of stated preference technique are as follows:

Contingent Valuation (CV)

CV methods employ a questionnaire format where respondents are asked how much they would be WTP or WTA for a specified gain or loss of a given good or service. Economic value estimates yielded by CV surveys are 'contingent' upon the hypothetical market situation that is presented to respondents and allows them to trade off gains and losses against money. WTP/WTA questions may be asked in a number of ways, including an open-ended format where the respondent is simply asked to state their maximum WTP/WTA, and a dichotomous choice format, where the respondent is required to answer yes or no to a 'bid' (e.g. are you willing to pay €x?). Although this method is considered to be controversial in some quarters, the contingent valuation method has gained increasing acceptance in recent years amongst many academics and policy makers as being a versatile and powerful methodology for estimating the monetary value of the non-market impacts of projects and policies.

An example of a CV study that is directly relevant to marine/coastal issues is Georgiou et. al. (1998). This study asks respondents what they are WTP to reduce the perceived risk of falling ill after bathing at two beaches with differing water quality in East Anglia in the UK. The survey asked the question, "what is the maximum amount of money that you would be willing to pay per year in the form of higher water rates to ensure that the bathing water at this beach passes the EC standard (does not fall below the EC standard)". Results showed that over the whole sample the mean WTP was £12.32 and £14.64 per year for the two study sites.

Advantages of CV:

- can estimate use and non-use values;

- a widely used and much researched environmental valuation technique;
- applicable to a wide range of ecosystem goods and services.

Disadvantages of CV:

like many questionnaire techniques can suffer from a wide range of biases.

Questionnaires need to be very carefully designed and pre-tested;

- very resource intensive. Reliable surveys need large sample sizes and hence consume manpower and finances;
- depending on the bid format used can be statistically complex to analyse.

Other issues:

- Most reliable when used to estimate the value of environmental gains and where the good or service of concern is reasonably familiar to respondents.

Choice Modelling (CM)

CM approaches involve respondents making choices between goods which are described in terms of their various attributes, offered in different amounts, or levels. There are two main choice formats: contingent ranking and choice experiments. In a contingent ranking exercise, respondents rank a set of alternative scenarios of good or service provision in order of preference. In a choice experiment, exercise respondents are presented with a series of scenarios along with their associated costs or prices and asked to choose their most preferred option. Survey results are then analysed statistically to arrive at the values of WTP that correspond to each scenario.

See Luisetti et al. (2008b) for an example of a choice experiment applied to coastal management and saltmarsh valuation.

Advantages of CM:

- as above for CV;
- more flexible than CV as it enables the attributes of an environmental gain scenario to be valued rather than just the overall scenario;

Disadvantages of CM:

- as above for CM, but even more attention needs to be paid to design issues and analysis can be even more complicated.

Revealed preference methods

Revealed preference methods infer individuals' preferences by observing their behaviour in markets in which a given environmental good is indirectly purchased. These approaches are reliant upon the assumption that non-market use values are indirectly reflected in consumer expenditure. Note that while these methods are grouped under the same overall category they differ

in having slightly different conceptual bases and in being applicable to the valuation of different environmental resources.

The Travel Cost Method (TCM)

The TCM enables the economic value of recreational use (an element of direct use value) for a specific site to be estimated. The method requires that the costs incurred by individuals travelling to recreation sites - in terms of both travel expenses (fuel, fares etc.) and time (e.g. foregone earnings) – is collected. The basic assumption is that these costs of travel serve as a proxy for the recreational value of visiting a particular site.

An interesting application of the TCM is described in Font (2000). The study applies the TCM to international tourist visits to a set of 10 protected natural areas in Mallorca. The results obtained from the model allows Fine to predict that over the course of a year tourists would be WTP a lower-bound figure of 30.21 billion pesetas (in 1997) for the option of being able to visit these sites.

Advantages of TCM:

- a well established technique;
- based on actual observed behaviour.

Disadvantages of TCM:

- can only estimate use values;
- really only applicable to specific sites (usually recreational sites);
- difficult to account for the possible benefits derived from travel, multipurpose trips and competing sites;
- very resource intensive. Reliable surveys need large sample sizes and hence consume manpower and finances;
- statistically complex to analyse.

Hedonic Pricing (HP)

HP may be applied to the valuation of environmental goods such as landscape amenity, air quality, and noise. The technique involves isolating the effect of these services on the demand for a marketed good. In most cases price data from the housing market are used. Analysis of the data estimates the implicit price which individuals are willing to pay for the relevant environmental characteristics. By trading these market goods, consumers are thereby able to express their values for the intangible goods, and these values can be uncovered through the use of statistical techniques. This process can be hindered, however, by the fact that a market good can have several intangible characteristics, and that these can be collinear. It can also be difficult to measure the intangible characteristics in a meaningful way.

The HP method has been mainly applied to data from housing and labour markets and especially the former with respect to valuation of environmental attributes. Research has been carried that has studied the effect on housing prices of proximity to landfill sites, or to aircraft noise, or air pollution. Leggett and Bockstael (2000) use HP to estimate the effect on waterside property

prices of a reduction in faecal coliform counts in Chesapeake Bay in the USA. Their results suggest that the increase in property price associated with this reduction in pollution amounts to up to 2% of average overall property value.

Advantages of HP:

- a well established technique;
- based on actual observed behaviour and (usually) existing data.

Disadvantages of HP:

- can only estimate use values;
- really only applicable to environmental attributes likely to be capitalised into the price of housing and/or land;
- confined to cases where property owners are aware of environmental variables and act because of them;
- market failures may mean that prices are distorted;
- data intensive and appropriate data may be difficult to obtain;
- statistically complex to analyse.

Averting behaviour and defensive expenditure

These approaches are similar to the TCM and HP, but they differ as they use as a basis individual behaviour to avoid negative intangible impacts as a conceptual base. For example, people buy goods such as safety helmets to reduce accident risk, and double-glazing to reduce traffic noise, and in doing so reveal their valuation of these bads. However, the situation is complicated (again) by the fact that these market goods might have more benefits than simply that of reducing an intangible bad. Averting behaviour occurs when individuals take costly actions to avoid exposure to a non-market bad (which might, for instance, include additional travel costs to avoid a risky way of getting from A to B). Again, we need to take account of the fact that valuing these alternative actions might not be a straightforward task, for instance, if time which would have been spent doing one thing is instead used to do something else, not only avoiding exposure to the non-market impact in question, but also producing valuable economic outputs.

Advantages of averting behaviour:

- has a sound theoretical basis;
- uses data on actual expenditures and data requirements can be modest;

Disadvantages of averting behaviour:

- not a widely used methodology;
- can only estimate use values;
- limited to cases where households spend money to offset environmental hazards/nuisances;
- confined to cases where those affected are aware of the environmental issue and act because of them;
- appropriate data may be difficult to obtain.

Cost of illness and lost output

Finally, methods based on cost of illness and lost output calculations are based on the observation that intangible impacts can, through an often complex pathway of successive physical relationships, ultimately have measurable economic impacts on market quantities. Examples include air pollution, which can lead to an increase in medical costs incurred in treating associated health impacts, as well as a loss in wages and profit.

Advantages of cost of illness and lost output:

- theoretically sound;
- very useful where there is a clearly established exposure-response relationship;
- can be a relatively simple exercise where exposure-response relationships have already been established and data on exposure and response is available;

Disadvantages of cost of illness and lost output:

- can only estimate use values;
- uncertainty regarding exposure-response:
 - are there threshold levels before damage occurs?
 - are there discontinuities in the exposure–response relationship?
- market failures may mean that the prices of market impacts are distorted;
- can be a very complex and resource intensive exercise where exposure-response relationships have not been established and where data on exposure and response is not readily available;

Pricing Approaches

Market prices

Market Prices data from ecosystem goods that are traded, either in local or international markets, offer perhaps the most visible indication of value. Products such as fish and shell fish are obvious examples. However, it may be necessary to adjust prices to account for government subsidies or taxes in order to obtain real or so called shadow prices.

Advantages of market prices:

- relatively simple;

Disadvantages of market prices:

- can only estimate direct use values;
- prices can be distorted by market failure;
- all pricing approaches are only a partial measure of value.

Opportunity cost/damage costs avoided

The Opportunity Cost approach estimates the benefits that are foregone when a particular action is taken. For example, storing carbon in managed ecosystems such as salt marshes can be 'valued' in terms of the damage costs avoided from the carbon emissions. In the strictest sense, opportunity cost should be viewed as the next best alternative use of a particular resource. Also opportunity cost allows estimation of the net value of a particular resource.

Advantages of opportunity cost:

- can be relatively simple;
- can be very useful where a policy precludes access to an area – for example estimating forgone money and in-kind incomes from establishment of a protected area.

Disadvantages of opportunity cost:

- can only estimate direct use values;
- may require detailed household surveys to establish economic and leisure activities in the area in question;
- all pricing approaches are only a partial measure of value.

Replacement costs

The replacement cost (or substitute goods) approach entails estimating the provision of an alternative resource that provides the function of concern. A wetland that provides protection against flooding could, for example, be valued, at the very least, on the basis of the cost of building man-made flood defences of equal effectiveness.

Shadow Project Costs consider the cost of providing an equal alternative environmental good at an alternative location. Such an approach may also be termed as a 'replacement cost' approach, which measure environmental value by applying the cost of reproducing the original level of benefit.

Advantages of replacement costs:

- can be relatively simple;

Disadvantages of replacement costs:

- can only estimate direct use values;
- all pricing approaches are only a partial measure of value.

Benefit transfer: generalizing results from existing environmental valuation studies

Environmental benefits transfer is a technique in which the results of previous environmental valuation studies are applied to new policy or decision-making contexts. In the literature, benefits transfer is commonly defined as the transposition of monetary environmental values estimated at one site (study site) to another site (policy site). The study site refers to the site where the original study took place, while the policy site is a new site where information is needed about the monetary value of similar benefits.

In the field of environmental valuation, benefits transfer has been applied extensively in various contexts, ranging from water quality management (e.g. Luken et al., 1992) and associated health risks (e.g. Kask and Shogren, 1994) to waste (e.g. Brisson and Pearce, 1995) and forest management (e.g. Bateman et al., 1995). Costanza et al. (1997) have extrapolated the monetary values of existing valuation studies to the flow of global ecosystem services and natural capital, and have thereby raised a number of questions as well as heavy criticism about the validity and reliability of benefits transfer.

A number of criteria have been identified in the literature for benefits transfer to result in reliable estimates (e.g. Desvousges et al., 1992; Loomis et al., 1995). These are summarised in Brouwer (2000):

- sufficient good quality data
- similar populations of beneficiaries
- similar environmental goods and services
- similar sites where these goods and services are found
- similar market constructs
- similar market size (number of beneficiaries)
- similar number and quality of substitute sites where the environmental goods and services are found.

Study quality is an important criterion, which can be assessed in a number of ways (Swedish Environmental Protection Agency, 2006). Above all, one can look at the internal validity of the study results, i.e. the extent to which findings correspond to what is theoretically expected. This internal validity has been extensively researched over the past three decades in valuation studies.

Studies should contain sufficient information to assess the validity and reliability of their results. This refers, among others, to the adequate reporting of the estimated WTP function. The reporting of the estimation of the WTP function should also include an extensive reporting of statistical techniques used, definition of variables and manipulation of data.

The most important reason for using previous research results in new policy contexts is that it saves a lot of time and money. Applying previous research findings to similar decision situations is a very attractive alternative to expensive and time consuming original research to inform decision-making.

In practice, several approaches to benefits transfer can be distinguished, which differ in the degree of complexity, the data requirements and the

reliability of the results. In principle, these approaches are all related to the use of either average WTP values or WTP functions. The first approach is most frequently applied, as it requires relatively little data or expertise, and is not very time consuming.

A first approach is where the unadjusted mean WTP point value is used from another study to predict the economic value of the benefits involved at the policy site. Ideally, this study focuses on the same environmental goods or services, but was carried out at a different location or at the same location at a different point in time.

A second approach is to use and average the unadjusted mean WTP estimates from more than one study, if available, instead of using the result from one study only. These are the two most frequently applied approaches to benefits transfer in practice. They are relatively data extensive and not very time consuming. However, although a quick and cheap alternative, especially compared to original valuation research, the results may be unreliable if circumstances and conditions in the new decision-making context in which they are used are very different from the ones prevailing in the original research.

A third approach is to use one or more mean WTP values adjusted for one or more factors which are, often based on *expert judgement*, expected to influence the value estimates at the policy site. For instance, mean WTP is sometimes adjusted for differences in income levels at the study and policy site, based on existing information about the income elasticity of WTP for the good or service in question, usually taken from the estimated WTP function in the original study.

A fourth approach is to use the entire WTP function from an original study to predict mean WTP at the policy site. Whereas the three previous approaches are referred to in the literature as 'unit value' or 'point estimate' transfers, this fourth approach is usually called 'function transfer'. The estimated coefficients in the WTP function are multiplied by the average values of the explanatory factors in the new policy context to predict an adjusted average WTP value. It has been argued that the transfer of values based on estimated functions is more robust than the transfer of unadjusted average unit values, since effectively more information can be transferred (Pearce *et al.*, 1994). However, this approach is usually more data intensive than the first three as information about all the relevant factors have to be readily available or collected.

A fifth approach is to use a WTP function, which has been estimated based on the results of various similar valuation studies. The difference between this approach and the fourth approach is that the WTP function is in this case estimated on the basis of either the summary statistics of more than one study or the individual data from these studies. In the literature, this approach is usually referred to as meta-analysis. Formally, meta-analysis is defined as the statistical analysis and evaluation of the results and findings of empirical studies.

Finally a sixth approach can be identified. That is the use of a value function - either one which was estimated in a single previous study (fourth approach) or one which was estimated based on multiple previous studies (fifth approach) - in which the coefficient estimates are adjusted when transferring the estimated value function to a new policy context based on prior knowledge. This approach corresponds to a more Bayesian oriented approach to benefits transfer

The fourth and fifth function approaches assume that the estimated coefficients remain constant, through time, across groups of people and across locations. However, based on previous knowledge and *expert judgement*, for instance from previous research at similar study sites or previous research at the new policy site, one may find a reason to adjust coefficient estimates. For example, available information about increases in income level in an area and available information about previously estimated income elasticities of WTP at different income levels, the coefficient estimate in the value function can be modified to better fit the new situation. This approach is expected to become especially relevant when functions are used in benefits transfer exercises, which were estimated a long time ago. Obviously, preferences reflected in stated WTP change as a result of changing circumstances. The fifth and sixth approach can be referred to as an 'adjusted function' approach, because in both cases a new WTP function is used, either based on the adjusted original function or a re-estimated function in a meta-analysis of multiple studies.

Thus, while benefit transfer provides a quick and cheap alternative to original valuation research, some conditions must be met if it should provide reliable results. Above all, the local circumstances and conditions in the new decision-making context need to be close enough to the ones prevailing in the original research. The risk of obtaining misleading results may be controlled and reduced by integrating more explaining variables into the transfer, however this also increases the data requirements and the complexity of the analysis. Also, the possibilities of conducting a sound and reliable benefits transfer hinge on the number, quality and diversity of valuation studies available – the larger, the better and the more diverse the existing set of studies is, the more likely will there be a primary study that is “close enough” to the policy site for results to be transferable.

Methods for eliciting non-economic values

There may be occasions where economic valuation is either not appropriate or not possible. This could be due to the nature of the ecosystem good or service, the degree of uncertainty surrounding environmental change, or because of objections to monetary valuation from stakeholders and/or the researchers involved in the study. In this situation a variety of qualitative valuation methodologies can be undertaken. Some of these are briefly summarised below:

Focus groups, In-depth groups. Focus groups aim to discover the positions of participants regarding, and/or explore how participants interact when discussing, a pre-defined issue or set of related issues. In-depth groups are similar in some respects, but they may meet on several occasions, and are much less closely facilitated, with the greater emphasis being on how the group creates discourse on the topic.

Citizens' Juries. Citizens' juries are designed to obtain carefully considered public opinion on a particular issue or set of social choices. A sample of citizens is given the opportunity to consider evidence from experts and other stakeholders and they then hold group discussion on the issue at hand.

Health-based valuation approaches. The approaches measure health-related outcomes in terms of the combined impact on the length and quality of life. For example, a quality-adjusted life year (QALY) combines two key dimensions of health outcomes: the degree of improvement/deterioration in health and the time interval over which this occurs, including any increase/decrease in the duration of life itself.

Q-methodology. This methodology aims to identify typical ways in which people think about environmental (or other) issues. While Q-methodology can potentially capture any kind of value, the process is not explicitly focused on 'quantifying' or distilling these values. Instead it is concerned with how individuals understand, think and feel about environmental problems and their possible solutions (Stagl, 2007).

Delphi surveys, systematic reviews. The intention of Delphi surveys and systematic reviews is to produce summaries of expert opinion or scientific evidence relating to particular questions. However, they both represent very different ways of achieving this. Delphi relies largely on expert opinion, while systematic review attempts to maximise reliance on objective data. Delphi and systematic review are not methods of valuation but, rather, means of summarising knowledge (which may be an important stage of other valuation methods). Note that these approaches can be applied to valuation directly, that is as a survey or review conducted to ascertain what is known about values for a given type of good.

Annex 3 Most Common Marine and Coastal Ecosystem Benefits/Services Valuation: case studies

Introduction

A number of countries are responsible for the North-East Atlantic Sea: Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, The Netherlands, Norway, Portugal, Spain, Sweden, Switzerland and United Kingdom. Although distant from the N-E Atlantic, Finland, Luxembourg and Switzerland are responsible for the river catchments that flow in the North-East Atlantic Sea.

Some of these countries (Sweden, the Netherlands and Belgium) have recently reported on the availability of economic valuation studies in the North-East Atlantic Sea and related coastal zones. These findings indicate that most of the studies were limited to the investigation of marginal costs rather than marginal benefits. The report also highlighted the need for new international studies valuing the benefits of nutrient loads reductions in the Baltic Sea (possibly including other countries such as Estonia, Latvia, Lithuania, Poland and Russia) and other areas within the N-E Atlantic. Fisheries production (commercial or recreational) could be valued in a more comprehensive way via the ecosystem services approach.

In this Annex, we first summarise the results of the report by Sweden, The Netherlands, and Belgium. Then, we report a table summarising the financial and economic valuation literature encompassing marine and coastal benefits in.

Member Country reports results

Sweden

The Swedish Environmental Protection Agency produced a document in spring 2008 covering all the Baltic Sea countries: Denmark, Estonia, Finland, Germany, Latvia, Lithuania, Poland, Russia, and Sweden. Each country carried out a literature review to investigate the existence of economic studies estimating the economic value of changes in the marine environment of the Baltic Sea. Other pressure/impact issues relating to eutrophication, fisheries, oil and marine debris, (off-shore) windmill parks, have also been evaluated.

Denmark: 17 studies are reported, few of which are Danish studies using economic methods. Most of the studies relate to the Baltic but use benefits transfer to get at country specific results.

Estonia: 6 international studies; 2 Estonian studies. All studies reviewed except for one, use data from the 1990s. Since the economic situation in

Estonia has changed significantly in the last decade, new updated studies are needed.

Finland: 12 studies discussing the value of water systems relating directly to the Baltic Sea in Finland were found. Some studies use data from 2000s onwards, others from the 1990s. However 5 ongoing valuation studies were identified showing a growing interest on the valuation of the benefits of the Baltic Sea ecosystem. Other studies (most of them ongoing studies) related to the inland waters of Finland.

Germany: 5 international existing studies concerning the economic valuation of the ecosystem services of the Baltic; 4 studies that focus on Germany for the valuation of the benefits in the Baltic; 6 work in progress (international and country specific) studies; 12 studies concerning water systems or coastal regions; 5 studies on tourism with some economic aspects (not specific economic investigations).

Latvia: 4 international studies on the economic valuation of the benefits of a cleaner Baltic; no Latvian studies on the Baltic were found; 2 inland water studies (one with 1996 data, the other with 2006 data).

Lithuania: 2 international studies; 4 specific Lithuanian case studies on water quality in rivers and Baltic coasts. Most studies are quite recent or carried out at the end of the 1990s.

Poland: 1 international study; 2 Polish studies (one in the 1990s and the other in 2004). Several studies which represent an extension of the results gathered in the three studies mentioned above.

Russia: 3 international studies that infer the value of Russian ecosystem services for the Baltic; no specific Russian studies for the Baltic were found.

Sweden: 3 international studies; 19 Swedish studies. All of them address a specific ecosystem service economic valuation in Swedish territory with respect to the Baltic: invasive species, eutrophication, fisheries, oil and marine debris, windmill parks.

A number of studies present some economic data related to stated preference values such as contingent valuation method (CVM) or choice experiment (CE) as well as revealed preference methods such as travel cost method (TCM). Sometimes the cost-effectiveness method is used. However, since it is difficult to use benefit transfer from one country to the other because of different socio-economic conditions or because the studies refer to very specific areas, to propose a unified value for the ecosystem services in the Baltic is problematic. Also, the former transition economies have none or few studies on the ecosystem services in the Baltic. A specific integrated international study on the values of the benefits of the ecosystem services of the Baltic is advocated.

The summary Table 3A of the Swedish report below highlight the major gaps found in the economic valuation of the ecosystem services of the Baltic.

Table 3A Ecosystem services coverage: what has been done and within which areas are further studies suggested in the country reports?

Category		Service	Degree of coverage in economic research, relative to other ecosystem services (aggregate based on number of related studies and the degree to which the specific services have been studied).	Degree of priority, regarding future studies
Supportive services	S1	Biochemical cycling	Low	?
	S2	Primary production	Low	?
	S3	Food web dynamics	Low	?
	S4	Diversity	Medium	!
	S5	Habitat	High	!!
	S6	Resilience	Low	?
Regulating services	R1	Atmospheric regulation	Low	?
	R2	Regulation of local climate	Low	?
	R3	Sediment retention	Low	?
	R4	Biological regulation	Low	?
	R5	Pollution control	Low	?
	R6	Eutrophication mitigation	Low	?
Provisioning services	P1	Food	High	!!!
	P2	Inedible resources	Low	?
	P3	Genetic resources	Low	?
	P4	Chemical resources	Low	?
	P5	Ornamental resources	Low	?
	P6	Energy	Low	?
	P7	Space & waterways	Low	?
Cultural services	C1	Recreation	High	!!!
	C2	Aesthetic value	High	!!!
	C3	Science & education	Low	?
	C4	Cultural heritage	Medium	!
	C5	Inspiration	Low	?
	C6	The legacy of nature	Medium	!

Legend:

? = The country reports don't specifically mention this service as an important priority for future research

! = Important area for further studies

!! = Very important area for further studies

!!! = Crucial area for further studies

The Netherlands

The Dutch have several studies in progress relating to the Baltic Sea, most of them to be finalised by the end of 2009:

- The economic importance of the Dutch part of the North Sea (Waterdienst);
- The cost of degradation of the marine environment (Steendam, N.);
- Possible applications of a Cost Benefit Analysis in the Marine Framework Directive (Sterk Consulting, Erasmus Universiteit);
- Ecological knowledge gaps relevant for performance of a Cost Benefit Analysis in the Marine Framework Directive (Arcadis);
- Exploration of the social analysis for the Marine Strategy Framework Directive (MSFD) (Witteveen+Bos);
- Several reports (in Dutch) on the economic analysis for the European Water Framework Directive.

It seems that The Netherlands followed the same line as the Swedish Environmental Protection Agency analysing the economic role of The Netherlands in the Baltic, as well as their responsibility for the Baltic Sea degradation and as analysis of the most relevant ecological knowledge gaps for CBA applications.

Belgium

The project 'Balancing Impacts of Human Activities in the North Sea' (BALANS) organised by Scientific Support Plan for a Sustainable Development Policy (SPSD II) investigates how to apply sustainable management in the Belgian part of the North Sea. BALANS attempts to bridge the gap between scientific data, information and application of knowledge in support of a sustainable management of the marine environment.

The BALANS report focuses on sand and gravel extraction, fisheries and shrimp farming. A model linking the ecology to the costs involved in those activities is presented. The figures presented come from a financial analysis of the returns of the activities investigated (sand and gravel extraction and shrimp fisheries). As such, the BALANS report does not present an economic valuation of the benefits of the ecosystem services in the Baltic. The final aim of the report was 'to create a decision support system with which decision makers and stakeholders could compare different policy options and choices against an array of ecological and socio-economic indicators'.

Literature Review of Marine and Coastal Ecosystem Benefits Valuation Studies

Based on the studies described in the Swedish Environmental Protection Agency report on the economic value of the ecosystem services provided by the Baltic Sea, and on other studies published in the literature, this section reports on the valuation of the marine and coastal ecosystem services for most of the North-East Atlantic Sea countries.

In Section 4 of this report we presented an Ecosystem Services Approach and an illustration of appropriate economic valuation of the benefits provided by marine and coastal ecosystems. We also illustrated how the economic value of some benefits can, in some cases, be approximated (often underestimating it) using financial valuation (market prices methods, actual costs, revenues, etc). Following that framework, we have summarised the most recent literature on marine and coastal ecosystem valuation in Table 4A. The table contains the following columns: firstly the benefits under investigation are presented; in the second column the ecosystem (marine and/or coastal) the benefits refer to is identified; the third column shows the most common ecosystem services related to the benefits under investigation (see Section 4.3 for details); fourth and fifth columns show the country and the region of the world within which the study took place (e.g. UK, Europe); the following columns provide a summary of the study, the economic or financial valuation results, the year the data were collected, and the reference for the study. The table is organised in alphabetical order by country. The table does not provide an exhaustive list, but reports the most relevant scientific and 'grey' literature studies. The problem of double counting may well be present in some studies and Table 5 in the Swedish report may need to be revisited in order to separate into intermediate and financial ecosystem service categories and related benefit values.

Following the approach of Table A4.2, we summarise below all the studies included in the table. On the basis that any given study may have addressed different issues and investigated the value of more than one benefit, we organised the section by single benefits estimated in the studies and related services: food provision (e.g. fish, shellfish); amenity (e.g. visual impacts) and recreation (e.g. sea angling, walking, bird watching) / cultural heritage, which can encompass the contribution of services such as water quality (reducing eutrophication), habitat, biodiversity conservation; raw materials; resilience and resistance; bioremediation of waste; disturbance prevention (e.g. flood protection).

Because the valuation of the benefits is context dependent, sometimes a specific ecosystem service can be estimated on its own right (e.g. water quality for a person requiring drinking water). In other situations, however, it can be included in the valuation of other benefits (following the previous example, water quality might be considered a service in a study valuing the benefit of recreational fisheries or food (fish) provisioning).

In each benefit (food provision, amenity and recreation / cultural heritage, carbon storage, raw materials, other ecosystem benefits/services) valuation sub-section we present first the case studies that report economic values; financial valuations of the benefits follow. A special sub-section is dedicated to the complex issue of oil spills.

Food provision

Several of the studies reviewed estimated the value of food provision (fisheries), but not all of them present economic values.

Economic value estimates

Luisetti PhD thesis (2008a) estimated the value of fish nurseries in the Blackwater (Essex – UK) with a production function method (Luisetti et al. 2008b), at £7.43/ha in 2008. The value of food provision is included in the estimation of the overall value of the total Polish coastal ecosystem, which is quantified on the basis of the Costanza et al. (2007) study at 76.76 MUSD in Martinez et al (2007). The economic value (WTP to preserve) of the Russian coastal forest of Kurshskaya spit, which then includes the value of food provision, is estimated at €0.7MEUR/year (values converted in 2007). For Sweden, a benefit transfer study (values taken from Eggert and Olsson, 2003) estimates the WTP for a cod-moratorium in the Kattegatt-Skagerrak area at €27-152 (values converted in 2007) – the values are mixed with recreational fishing.

Financial value estimates

Beaumont et al. (2007) report a literature review of several studies which address the valuation of food provision, but only those for the Banco D. Joao de Castro (Atlantic Ocean) and Belgium (North Sea) report financial values: €1.5 million/year in the former case; and €91,911,000 for the Belgian sea fishing industry in 2002. The study of Vetemaa et al. (2003), estimates the financial value (converted in Euros in 2007) of safe catch fishing (catch that avoids bycatch of rare and endangered water birds) in Estonia for 1998-99: €137.80-206.69 (thousand). Some financial values are available in the Kaliningrad regional public Fund “21st century” Russian report on the estimated value of fish resources over a hundred year period (starting in 1999) in the Kurshskaya spit: €75 MEUR (converted values in 2007). The same report estimates the value of food (wild fruit and mushrooms) produced in the Russian coastal forest of Kurshskaya spit: €100/year (values converted in 2007). In Gren et al. (2007), the value of food provision in Sweden is approximated by considering the costs involved in the control for invasive species: €18-45 MEUR.

Amenity & Recreation / cultural heritage

Most of the studies in the literature of ecosystem services/benefits, as highlighted in the Swedish report, value the amenity and recreation of marine and coastal areas. This is an interesting benefit to value, because if we consider that people go to an area to enjoy the beauty of nature there, we see that the amenity and recreation benefit includes the contribution provided by the overall ecosystem (e.g. nutrient cycling, regulation of water flow and quality, habitat, biodiversity conservation etc.).

Economic value estimates

Using a choice experiment, in which the good under investigation (e.g. saltmarshes) can be decomposed and analysed in its components or attributes, Luisetti et al. (2008a) found that the marginal WTP (MWTP) for

saltmarshes in the Blackwater estuary (Essex – UK) is £4.31 (use value) and £3.57 for non-use benefits. The aggregate value of the amenity and recreation for the salt marshes in the whole estuary was calculated for three different scenarios and two different policies; the most conservative figure is: £4.4 million.

Some studies (Landeberg and Dubgaard, 2007; Landeberg, 2007, 2008) have focused on amenities in Danish coastal areas while investigating the impact of off-shore wind farms. In the first two studies, a WTP/km/year to move the wind farms further away from the coast is estimated: in the interacted model the WTP decreases with distance. The maximum WTP is €70 when the farms are at 15 km from the coast, and the minimum WTP is €0 when they are 50 km from the coast, in the first study; in the second study, three distances are presented (12, 18, 50 km) and the min-max WTP is €31.6-154.9. The visual impact of wind farms is also estimated by Liljestam and Söderqvist (2004) for Sweden (Björkö – east coast) as a mean WTP (€36-75 in 2007 values) for different scenarios involving land based and off shore (different distances) wind farms.

Recreational fishing is reported for different areas in different studies. Jensen et al. (2002) present the WTP for the occasional and the passionate angler: €69 in the first case, and € 198, in the second (values are converted in 2007). Toivonen et al. (2004) estimate the WTP per fisherman, per year of €83 (values converted in 2007), and the aggregate value in €93 million. Fiskeriverket (2008) report a total WTP estimate (€60 million) for recreational fishing in Sweden. In more specific areas of Sweden like Bohus (south-west) Paulrud (2004) estimates the compensating variation (marginal value) of sport-fishing (€0.6-1.0 per number of catches, €1.3-1.8 per kilo – data of 1998 converted in 2007 monetary values), and for the Stockholm archipelago the total WTP in case of doubled catch is estimated to be €0.42 million (from 2002 data into 2007 monetary values) by Souturkova and Söderqvist (2005).

Some studies (Toivonen et al. 2000; Toivonen et al. 2004; Parkkila et al. 2005) give the valuation of the marine and/or catchment areas as it is not always easy to distinguish which ecosystem actually supports the fisheries (for example salmon). Parkkila et al. (2005), for example, investigate with a contingent valuation study the value of increasing salmon catch in the river Simojoki (Finland) which empties in the Gulf of Bothnia (The Baltic Sea). The WTP estimate results are: €50-56/fisherman/fishing season (in 2007 values); and the aggregate value is at €31,000/year. Whereas Toivonen et al. 2004 used the contingent valuation method (in 1999) to estimate the value for preserving current fisheries in Finland. The result of that study was an aggregated WTP at €180 million.

Other studies report the values of general amenity and recreation. Povilanskas et al. (1998) estimate the value of amenity and recreation for the Matsalu Bay in Estonia with several methods: WTP, in the range €71.3-1715.5 (thousand); and consumer surplus (using travel cost data), which is estimated to be €83.9 (thousand). The WTP to preserve Rügen Island in Germany is estimated by Degenhart and Groemann (1998) in €0.45/per night. In Russia,

the Kaliningrad regional public Fund “21st century” reports the consumer surplus valuation of travelling to the Kurshskaya spit (in 1999) is €1.7-2.9 (million) in 2007 monetary values. Laholm Bay in the west-coast of Sweden was valued by Fryblom (1998) with a contingent valuation study. The mean annual WTP is (in 2007 values) €90, and the total annual WTP is €10.8 (million). Povilanskas et al. (1998) estimated the WTP and the consumer surplus for two specific areas in Latvia: the Curonian Lagoon and the Nemunas delta. For the first, the WTP ranges are €1.1-19 (in 2007 values), and the consumer surplus (travel cost data) is €13,640. For the delta, WTP ranges are €1.1-17.3, and the consumer surplus is €277.

Amenity and recreation are also estimated when water quality is improved because of a reduction in nutrients load (reducing eutrophication). This implies the indirect valuation of water quality for recreational uses. For Finland, there are two main studies. Siitonen et al. (1992) using benefit transfer values from international studies estimate amenity and recreation for the period 1980-1989 to be (in 2007 values) €269,000-515,000/year. Kosenius (2004) estimates a WTP/person/year of €24.9 (€308,000 aggregate value) for recreation possibilities because of increased water quality and reduced risk of shell fish poisoning. In the Baltic Sea costs of Poland amenity and recreation are valued by Markowska and Zylicz (1996, 1999) with a contingent valuation study that results in a WTP/person/year that, depending on the choice presented and the sample, ranges between €4.42-137.36 (in 2007 values). The consumer surplus (estimated from travel data) per year for Laholm Bay in the west-coast of Sweden is €1.4-3.9 million (in 2007 values) as estimated by Sandstrom (1996). WTP values for the Stockholm archipelago were estimated by Söderqvist (2000), which obtains a median WTP/person/month of €6.0 (in 2007) values, and by Souturkova (2001), which simulates the values using travel costs data in a RUM model and obtains a recreational benefit (in 2007 values) of €10-307 (million). In France the mean WTP/household/year is (in 2007 values) €38.90 as estimated in 1995 by Le Goffe. In Scotland, Hanley et al. (2001) with a revealed-preference study estimated the benefit of increased water quality for the south-west to be €11.40/person/year. In the same area, Hanley and Kristrom (2002) estimated the WTP to be €19.44 for the city of Ayr and €11.84 for the city of Irvine. Another study related to water quality and reduction of gastroenteritis risk, Machado and Mourato (2002), provided a WTP/person of €47.92.

Other studies look at the value of water quality per se (drinking use, or biodiversity maintenance). Turner et al. (1999), for example, using 1995 data, estimated the value of water quality because of nutrient (eutrophication) reduction originally for Sweden and Poland. WTP estimates (See table A4.2) were then obtained for other countries transferring the values from Sweden – mean WTP €446-798 in 2007 values - (Denmark, Finland, Germany) and from Poland (Estonia, Latvia, Lithuania, Russia). Markowska and Zylicz (1999) estimate a contingent valuation for Sweden originally and, using benefit transfer values from that study, infer the WTP also for Finland, Germany, Latvia, Poland, Russia (see Table A4.2 for the different WTP values). More recently, Atkins and Burdon investigated the value of water quality for Denmark using a contingent valuation study: mean maximum

WTP/month/person for the limited period of 10 years is €12.85 (in 2007 values). For Sweden, Franzén et al. (2006) estimated the WTP of increased water quality using benefit transfer values from the Hökby and Söderqvist (2003) paper obtaining, for example (see Table A4.2 for details), an annual WTP of 77MEUR (in 2007 values) for 10,000 tons nitrogen loads reduction.

Financial value estimates

Financial valuations often cover recreational fishing values. A financial valuation of recreational fishing in the Danish Baltic coast is provided in COWI (2007): €1.75 billion. Recreational sailing is valued in VisitDenmark (2006) as an average spend per day, €36-38 (values of 2006 converted in euros in 2007), and as the turnover from sailing, €33 MEUR. Jensen et al. (2002) provide the operating costs for two categories of occasional and passionate angler: €110/year (occasional); €397/year (passionate). The total net benefits estimated are: €532,944. In Toivonnen et al. (2004) study, the mean fishing expenses are valued at €173/fisherman/year and the aggregate value is €227 million. The financial valuation of recreational fishing reported by the Federal Research Centre for Fisheries (Germany) is €2.85-7.65 million (in the years 2004-2005). Fiskeriverket (2008) present the total yearly turnover of recreational fishing in Sweden (€53 million).

Carbon storage

Luisetti et al. (2008b) estimated the benefit of carbon storage from saltmarsh recreation in the Blackwater estuary using different damage cost avoided values (in a range of £7-230) per annum.

Raw materials

The value of raw materials such as grassland/hay and reeds has been investigated by Gren et al. (1995) for the Baltic Sea region using market prices: grassland/hay is valued at €81.5-151.4 and reeds at €145.1-944.5 (in 2007 values).

Other ecosystem benefits/services valuation studies

The Beaumont et al. (2007) paper, address also resilience and resistance, bioremediation of waste, and disturbance prevention for some areas. However, neither economic nor financial values for those services are provided in the study.

O' Garra (2009) presents a first attempt at measuring bequest (non-use) values in developing countries (Muavuso peninsula –Fiji). The author measures with a contingent valuation method the bequest value of the 'iqoliqoli' (customary fishing rights for the villagers only). The WTP has been elicited in monetary and in time units. The results are: FJ\$1.25-1.41 (US\$ 0.69-0.73)/person/week; or FJ\$ 183.90 (US\$ 106.91)/household/year. Here the ecosystem service valued is future food provision (assured only if the

iqoliqoli will stay in the family). The study is reported here more for the method and the benefit (i.e. bequest value) interests as it is clearly not meaningful to transfer values data from this study context to the OSPAR region.

Oil spills: water quality value and costs of cleaning after an accident

The WTP to prevent oil spills in the Gulf of Finland has been measured using the contingent valuation method by Ahtiainen (2007). For a one-time payment, the results show a WTP of €28/person and an aggregate value of €112 (million). That gives indirectly the value for clean water and clean coasts.

Other studies have been carried out around Europe to assess the costs of oil spills in the Baltic Sea (see for example Hall (2000) for the UK, Ireland, Denmark, Sweden, Norway, Netherlands and Germany; Etkin (2000) for a worldwide investigation; and Forsman (2006, 2007) for Sweden. Although that information is very valuable, it only gives a partial insight into the real economic value of a clean and healthy coast, and since specific ecosystem services to be valued are not identifiable with this methodology, those studies have not been included in Table 4A.

Conclusions

This brief overview of the literature on marine and coastal ecosystem services and benefits shows how difficult it is sometimes to identify specific ecosystem services to be valued given the interrelationships within and between ecosystems over space and time. Our review focuses mainly on European studies including published papers and 'grey' literature, which indicates a growing interest in the issue of ecosystem services valuation. However, an agreed common framework to be used for economic valuation is still missing. That leads sometimes to double counting. In other contexts the analyst is forced, because of data and science gaps, to use financial data to approximate benefits values. Also, the literature in many cases was carried out in the late 1990s and new more structured studies are required if an adequate values database is to be established.

Table 4A European and Non-European case studies on marine and coastal ecosystem services valuation. In the reference column, we added the symbol (¿) for those case studies that are at risk of double counting.

BENEFIT	ECOSYSTEM	ECOSYSTEM SERVICES	COUNTRY	REGION	ISSUE ADDRESSED IN STUDY	ECONOMIC VALUATION	FINANCIAL VALUATION	YEAR OF DATA	REFERENCE
European case studies									
Food provision; resilience and resistance	Marine	Primary production; climate mitigation; regulation of water flow and quality; biodiversity maintenance	Banco D. Joao de Castro	Atlantic Ocean	Identification of ecosystem goods and services at specific locations based on the ecosystem approach of the Millennium Ecosystem Assessment (UNEP, 2006) to validate a theoretical Ecosystem Approach framework.		Fisheries: €1.5 Million per year.	Various years	Beaumont et al. (2007)
Food provision; bioremediation of waste; cultural heritage; amenity and recreation	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; climate mitigation; regulation of water flow and quality; refugia and nursery for many aquatic species; biodiversity maintenance	Belgium (North Sea)	Europe	Identification of ecosystem goods and services at specific locations based on the ecosystem approach of the Millennium Ecosystem Assessment (UNEP, 2006) to validate a theoretical Ecosystem Approach framework.		Belgian sea fishing industry (2002): €91,911,000.	Various years	Beaumont et al. (2007)

Food provision; carbon storage; bioremediation of waste; cultural heritage; amenity and recreation	Coastal areas	Sediment and nutrient cycling; primary production; water cycling; climate mitigation; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	Denmark-Germany (Lister Deep - North Sea)	Europe	Identification of ecosystem goods and services at specific locations based on the ecosystem approach of the Millennium Ecosystem Assessment (UNEP, 2006) to validate a theoretical Ecosystem Approach framework.		Various years	Beaumont et al. (2007)
Water quality	Coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality	Denmark (Randers Fjord - east coast of Jutland)	Europe	Economic valuation of water quality improvements. Method used: contingent valuation (WTP). Mail survey conducted in Denmark (Arhus County).	Mean maximum WTP per month €12.85 (2007) per person for the limited period of 10 years.	2003	Atkins and Burdon (2006)
Water quality	Coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality	Denmark (Randers Fjord - east coast of Jutland)	Europe	Economic valuation of water quality improvements. Method used: contingent valuation (WTP). Mail survey conducted in Denmark (Arhus County). This is an update of the Atkins and Burdon	Mean maximum WTP per month €8.2 (2007) per person for the limited period of 10 years.	2003	Atkins et al. (2007)

(2006) results. Data are now analysed with a decision tree analysis.

Amenity (visual impact off-shore wind farms)	Marine	Primary production; climate mitigation; regulation of water flow and quality; biodiversity maintenance	Denmark (eastern coast)	Europe	Economic valuation of offshore wind farms visual disamenities. The role of distance to shore Method used: choice experiment (WTP).	Range of distances considered: 10-50 km. Five different models estimated. WTP/km per household per year: min €0 at 50 km; max €70 at 15 km.	Landeburg and Dubgaard (2007)
Amenity (visual impact off-shore wind farms)	Marine	Primary production; climate mitigation; regulation of water flow and quality; biodiversity maintenance	Denmark (eastern coast)	Europe	Economic valuation of offshore wind farms visual disamenities. The role of distance to shore is examined. Method used: choice experiment (WTP). Extension of the Landeburg and Dubgaard (2007) results.	Three different samples and three distances (12, 18, 50 km): min and max WTP €31.6-154.9 (no year of reference indicated) per household, per year.	Landeburg (2007)
Amenity (visual impact on-land/off-shore wind farms)	Marine	Primary production; climate mitigation; regulation of water flow and quality; biodiversity maintenance	Denmark (eastern coast)	Europe	Preference between on-land and off-shore wind farms. Method used: choice experiment (probit model to show preferences).		Landeburg (2008)

Amenity & recreation	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	Denmark (Baltic Sea area of the country)	Europe	Estimate of tourism in the coastal regions of the Baltic Sea.	Tourism industry (2007): €1.75 billion; cruise tourism (2007): €43.4 million.	2006	COWI (2007)
Amenity & recreation (recreational sailing)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	Denmark (harbours)	Europe	Survey on expenditures for recreational sailing.	Sailor's average spent per day (2007): €36-38; turnover from sailing tourism (2007): €33 million.	2006	VisitDenmark (2006)

Recreation (recreational fisheries)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; refugia and nursery for many aquatic species; biodiversity maintenance	Denmark (west coast of Jutland)	Europe	Investigation on the revenues and WTP for angling.	Occasional angler (2007): the WTP is €69. Passionate angler (2007): the WTP is €198.	Occasional angler (2007): the operating cost/year is €110. Passionate angler (2007): the operating cost/year is €397. Total estimated net benefit from angling: €532,944.	2001	Jensen et al. (2002)
Recreation (recreational fisheries)	River	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; refugia and nursery for many aquatic species; biodiversity maintenance	Denmark and Scandinavian countries	Europe	Contingent valuation to estimate recreational fishing opening a new river to fishing. Denmark and Scandinavian countries surveyed. The use and non-use value to preserve the Nordic freshwater fish stock was also estimated.	Compensating surplus (in € in 2007 values) to estimate the use values of recreational fishing depending on the species caught. Salmon and sea trout in river: €138/year. Perch and pike-perch in lake: €111/year. Grayling, brown trout and arctic char in lake: €138. Preservation of Nordic freshwater stock: €300.			Toivonen et al. (2000)
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Estonia (Baltic Sea)	Europe	Benefit transfer values from the Lithuanian results of a contingent valuation to reduce eutrophication (improving water quality).	Values converted Euro in 2007: €41.6 per capita; €47.4 million (aggregate value)		1995	Markowska and Zylicz (1999)

Amenity and recreation	Marine and coastal areas (general nature, coastal meadows, floodplains, forested meadows)	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	Estonia (Matsalu Bay)	Europe	WTP and Consumer Surplus estimation of amenity and recreation in Matsalu Bay	WTP - median (thousand/2007): General Nature: €1715.5 (referendum); €71.3-238 (discrete choice); 952.1 (payment card); €762.3 (open ended). Consumer Surplus (travel cost) (thousand/2007): €83.9	1997	Povilanskas et al. (1998)
Food provision (fisheries)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	Estonia (Baltic Sea)	Europe	Valuation of safe catch of fishing (that fishing that avoids bycatch of rare and endangered water birds).	Values of safe catch (avoiding bycatch of rare and endangered water birds) of fish: €137.80-206.69 (thousand in 2007).	1998-99	Vetemaa et al. (2003)
Amenity & recreation (reduced eutrophication)	Coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for	Finland	Europe	Recreational values of increased water quality because of reduced amount of nutrients.	Benefit transfer values from international studies €(in 2007)/per year during the period 1980-1989: €269000-515000.	1980-1989	Siitonen et al. (1992)

		many aquatic species; refugia for birds; biodiversity maintenance							
Amenity & recreation (reduced eutrophication)	Coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	Finland (Hanko - Gulf of Finland)	Europe	Recreational values of increased water quality because of reduced amount of shellfish poisoning elements.	WTP (contingent valuation method) € (in 2007): €24.9/person/year. Aggregate tourist's WTP: €308000.		2003	Kosenius (2004)
Amenity & recreation (recreational fishing)	Coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	Finland	Europe	Economic valuation of recreational fishing (in streams and lakes) in the Nordic countries. WTP for recreational fishing and to preserve current fisheries stock. Financial valuation of the mean fishing expenses in Finland and the other Nordic countries.	WTP (contingent valuation method) € (in 2007): €83/fisherman/year. Aggregate value: €93 (million). Estimated value for preserving current fisheries: €180 (million)	Mean fishing expenses € (in 2007): €173/fisherman/year. Aggregate value: €227 (million).	1999	Toivonen et al. (2004)

Amenity & recreation (recreational fishing)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	Finland (Gulf of Bothnia)	Europe	Economic valuation of increasing salmon catch in the river Simojoki, which empties into the Gulf of Bothnia.	WTP (contingent valuation method) € (in 2007): €50-56/fisherman/fishing season. Aggregate value of increasing salmon catch: €31000/year.	2004	Parkkila (2005)
Amenity & recreation (preventing oil spills)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	Finland (Gulf of Finland)	Europe	Economic valuation (WTP - contingent valuation method) of amenity and recreation (including biodiversity values) of preventing oil spills in the Gulf of Finland.	WTP, one-time payment, € (in 2007): €28/person. Aggregate value: €112 (million).	2006	Ahtiainen (2007)
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Finland	Europe	Benefit transfer values from the Swedish results of a contingent valuation to reduce eutrophication (improving water quality).	WTP (benefit transfer - contingent valuation) € (in 2007): €369/year (assuming zero WTP for non respondents). Aggregate WTP: €1386 (million)/year.	1998	Turner et al. (1999)

Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Finland	Europe	Benefit transfer values from the Swedish results of a contingent valuation to reduce eutrophication (improving water quality).	Benefit transfer values from Swedish contingent valuation study result used to infer values for Finland. Mean WTP € (in 2007): €175/person. Aggregate value: €656 (million).	1999	Markowska and Zylicz (1999)
Amenity & recreation (water quality - bathing/health impacts)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	France	Europe	Investigation on the cost of eutrophication using a contingent valuation study. The WTP was elicited for the recreational use of water, and to prevent eutrophication (conserving the ecosystem).	Mean WTP € (in 1995)/household/year: €38.90 (recreation); €28.95 (conserving the ecosystem).	1995	Le Goffe (1995)
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Germany	Europe	Benefit transfer values from the Swedish results of a contingent valuation to reduce eutrophication (improving water quality).	WTP (benefit transfer - contingent valuation) € (in 2007): €384/year (assuming zero WTP for non respondents). Aggregate WTP: €933 (million)/year.	1998	Turner et al. (1999)
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of	Germany	Europe	Benefit transfer values from the Swedish results of a contingent valuation to reduce eutrophication	Mean WTP € (in 2007): €179/person. Aggregate value: €435 (million).	1999	Markowska and Zylicz (1999)

		water flow and quality.			(improving water quality).			
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Germany	Europe	Benefits valuation of reducing eutrophication in the Baltic Sea. Results based on Markowska and Zylicz (1999).	Benefit transfer values from Swedish contingent valuation study result used to infer values for Germany. Mean WTP/capita/year € (in 2007): €354; aggregated value €861 (million)/year.	1999	Söderqvist (2000)
Amenity & recreation (recreational fishing)	Coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	Germany	Europe	Financial valuations of actual recreational fishing catch in Germany (mainly in the Baltic Sea). Market prices method.	Aggregate value of recreational fishing in Germany: € (in 2004-2005) 2.85-7.65 (million).		Federal Research Centre for Fisheries (Bundesforschungsanstalt für Fischerei) (2007)
Amenity & recreation	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for	Germany (Rügen Island)	Europe	Economic valuation of nature conservation in Rügen Island. Contingent valuation method used.	WTP estimate: €0.45/per night	1998	Degenhardt & Gronemann (1998)

		many aquatic species; refugia for birds; biodiversity maintenance.						
Water quality (reducing pollution)	Coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality	Greece	Europe	Contingent valuation method used (open ended question) to assess the WTP (every four months for four years) to have a sewage treatment plant.	WTP to be paid every four months over four years: €16.84. Aggregate WTP: €208,294.55		Jones et al. (2008)
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Latvia	Europe	Benefit transfer values from the Polish results of a contingent valuation to reduce eutrophication (improving water quality).	WTP/capita/year/€ (in 2007): €131.	1995	Turner et al. (1995)
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Latvia	Europe	Benefit transfer values from the Lithuanian results of a contingent valuation to reduce eutrophication (improving water quality).	Mean WTP € (in 2007): €128/person.	1994	Markowska and Zylicz (1999)
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of	Latvia	Europe	Benefit transfer values from other country results of a contingent valuation study to reduce eutrophication	WTP/capita/year/€ (in 2007): €59.		Gren et al. (1997)

		water flow and quality.			(improve water quality).			
Amenity and recreation	Marine and coastal areas (general nature, coastal meadows, floodplains, forested meadows)	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	Latvia (Curonian Lagoon and Nemunas delta)	Europe	WTP and Consumer Surplus estimation of amenity and recreation in Curonian Lagoon and Nemunas delta.	WTP - median (thousand/2007): Curonian Spit - General Nature: €19 (referendum); €1.1-3.6 (discrete choice); €13 (payment card); €4.3 (open ended). Consumer Surplus (travel cost) (thousand/2007): €13640; Nemunas delta - General Nature: €17.3 (referendum); €1	1997	Povilanskas et al. (1998)
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Lithuania	Europe	Contingent valuation study to elicit WTP for an increase in water quality.	Median WTP/person/year € (in 2007): €8.04.	1994	Turner et al. (1995)
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Lithuania	Europe	Contingent valuation study to elicit WTP for an increase in water quality (upgrade landfill, sewerage, recycling programs).	WTP: for drinking water €0.02-0.08 (in 2007 values); for sewerage extension €0.08-0.57 (in 2007 values).	1999	Milieu Ltd. With AAPC (2001)

Food provision (fisheries); leisure and recreation; bequest and existence values	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	Mediterranean and Black Sea	Europe	Literature review on good and service valuations provided by The Mediterranean and Black Sea marine ecosystems.	Amenity & recreation: mean WTP in Venice €36 (non-use), €56 users, €35 potential users (Alberini et al., 2004); see also Vinci et al (2006) and Brau & Cao (2006). Bequest values: median WTP for the Mediterranean monk seal (in 1998) €12 (Langford et al., 1998).	Existing anchovy fisheries revenues in the Black Sea: €12.57 million.	Various years	Remoundou et al. (2009)
Food provision; raw materials; carbon storage; disturbance prevention (flood protection); bioremediation of waste; cultural heritage; amenity and recreation	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; climate mitigation; regulation of water flow and quality; habitat, refugia and nursery for many aquatic species and sea-birds; biodiversity maintenance	Poland-Russia (Gulf of Gdansk)	Europe	Identification of ecosystem goods and services at specific locations based on the ecosystem approach of the Millennium Ecosystem Assessment (UNEP, 2006) to validate a theoretical Ecosystem Approach framework.		Polish amber market: €20 million	Various years	Beaumont et al. (2007)
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and	Poland (Baltic Sea)	Europe	Contingent valuation study to elicit WTP for an increase in water quality reducing eutrophication in the Baltic Sea.	Mean WTP/person/year € (in 2007): €98.44 (dichotomous choice - random Polish citizens); €4.42 (open ended); €140.69 (payment card); €137.36 (dichotomous choice -		1994	Markowska and Zylicz (1996, 1999)

		quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance				Polish tourist on beaches).		
Food provision; raw materials; carbon storage; disturbance prevention (flood protection); bioremediation of waste; cultural heritage; amenity and recreation	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; climate mitigation; regulation of water flow and quality; habitat, refugia and nursery for many aquatic species and sea-birds; biodiversity maintenance	Poland	Europe	Total Polish coastal ecosystem services value based on compensating variation in Costanza et al. (1997).	Natural 'ecosystem services product': 76.76 MUSD.	1997	(¿) Martinez et al. (2007)
Amenity & recreation (water quality - bathing/health impacts)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	Portugal (Estoril coast)	Europe	Contingent ranking study to elicit the WTP of people to avoid gastroenteritis episodes because of poor water quality, and to investigate the benefits of moving from a poor and an average water quality to a 'blue flag' water quality.	WTP € (in 2002)/person: €47.92 (to avoid gastroenteritis); €19.89 (benefits of moving from a poor quality beach to a 'blue flag' beach); €8.46 (benefits of moving from average to 'blue flag').	2002	Machado and Mourato (2002)

Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Russia	Europe	Benefit transfer values from the Lithuanian results of a contingent valuation to reduce eutrophication (improve water quality).	Mean WTP € (in 2007): €31/person. Aggregate value: €262 (million).	1994	Markowska and Zylicz (1996, 1999)
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Russia	Europe	Benefit transfer values from the Polish results of a contingent valuation to reduce eutrophication (improve water quality).	WTP (benefit transfer - contingent valuation) € (in 2007): €116/person/year (assuming zero WTP for non respondents). Aggregate WTP: €426-841 (million)/year.	1998	Turner et al. (1999)
Food provision (fish); amenity and recreation (recreational fishing)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	Russia (Kurshskaya spit - Kurshskiy bay)	Europe	Valuation commercial and recreational fishing catch in Russia (Kurshskiy bay). Market prices method.	Estimated value of fish resources in Kurshskiy bay over 100 years (€ in 2007): €75 (million).	1999	Kaliningrad regional public Fund "21st century": http://www.biodat.ru/index_e.htm

Amenity and recreation	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	Russia (Kurshskaya spit - Kurshskiy bay)	Europe	Consumer surplus valuation of travelling to the Kurshskaya spit from different Russian cities.	Estimated value of recreational resources of the Kurshskaya spit National Park (€ in 2007/year): €1.7-2.9 (million).		1999	Kaliningrad regional public Fund "21st century": http://www.biodat.ru/index_e.htm
Food provision (wild fruit); carbon storage; energy provision (direct timber use); amenity & recreation.	Coastal wetlands	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many species; refugia for birds; biodiversity maintenance	Russia (Kurshskaya spit - Kurshskiy bay)	Europe	Financial valuation of the Kurshskaya spit forest resources and economic valuation to preserve the forest.	WTP to preserve the Kurshskaya spit National Park (values in € in 2007): €0.7 (million)/year.	Forest value as source of (values in € in 2007): fuel €24k/year; industrial timber €80k/ year; food €100/year; CO2 absorption €15.8 (million)/year	1999	Kaliningrad regional public Fund "21st century": http://www.biodat.ru/index_e.htm
Food provision (fish); energy.	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for	Sweden	Europe	Approximate value of fisheries and water energy affected by invasive species. Financial valuation assessing the costs of controlling for invasive species.		Costs of controlling for invasive species (€ in 2007): €18-45 (million)	2006	Gren et al. (2007)

		many species; refugia for birds; biodiversity maintenance						
Amenity and recreation	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Sweden (Laholm Bay - west coast)	Europe	WTP estimated by means of a contingent valuation study.	Mean annual WTP (€ in 2007): €90. Total annual WTP (€ in 2007): €10.8 (million).	1998	Frykblom (1998)
Amenity and recreation	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Sweden (Laholm Bay - west coast)	Europe	Consumer surplus values simulated using the travel-cost method, the household production function method, and a random utility maximisation (RUM) model.	Estimated increase in consumer surplus/year (€ in 2008): €29-65 (million) for a 50% reduction in the nutrient load along the entire Swedish coastline; €1.4-3.9 (million) for a 50% reduction of nutrient load in the Laholm Bay.	1996	Sandstrom (1996)
Amenity and recreation	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Sweden (Stockholm archipelago)	Europe	WTP values of recreational benefits improving water quality simulated using a random utility maximisation (RUM) model based on visits in the archipelago.	Recreational benefit value (€ in 2007): €10-307 (million).	1999	Soutukorva (2001)

Amenity and recreation	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Sweden (Stockholm archipelago)	Europe	WTP values of recreational benefits improving water quality using the contingent valuation method.	Median WTP € (in 2007)/person/month: €6.0. Aggregated value € (in 2007): €61-101 (million).		1998	Söderqvist and Scharin (2000)
Amenity & recreation (recreational fishing)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	Sweden	Europe	Financial calculation of recreational fishing and its dependant business. Consumer surplus on recreational fishing. WTP estimate on a scenario in which the catch possibilities would double.	Total consumer surplus € (in 2007): €80 (million). Consumer surplus per kilo fish € (in 2007): €4.2. WTP estimate € (in 2007): €60 (million).	Total yearly turnover (€ in 2007): €53 (million).	2007	Fiskeriverket (2008)
Food provision (fish)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many species; refugia for birds; biodiversity maintenance	Sweden (Skagerrak and Kattegatt)	Europe	Contingent valuation study to estimate the WTP for improved cod stock in Skagerrak and Kattegatt.	Mean WTP for improved cod stock € (in 2007): €25-99. Aggregate value € (in 2007): €77 (million).		2004	Olsson (2004)

Amenity & recreation (recreational fishing)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	Sweden (Bohus - south west)	Europe	Contingent valuation study to estimate the WTP of sport-fishing in Bohus.	Compensating variation results € (in 2007): €0.6-1.0 marginal value per number of catches; €1.3-1.8 marginal value per kilo.	1998	Paulrud (2004)
Amenity & recreation (recreational fishing)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	Sweden (Stockholm archipelago)	Europe	WTP values of recreational benefits improved recreational fishing is estimated using a random utility maximisation (RUM) model based on a mail survey interviewing sport-anglers and randomly selected inhabitants in the counties of Stockholm and Uppsala	Total WTP estimated in case of doubled catch in € (in 2007): €0.42 (million).	2002	Soutukorva and Söderqvist (2005)
Amenity & recreation (visual impact)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat	Sweden (Björkönen - Swedish east coast)	Europe	Valuation of alternative locations for windmills parks (in-land or off-shore). WTP is estimated using a contingent valuation method	Estimated mean WTP € (in 2007) for each scenario: A. €36; B. €83; C. €75. Aggregation: A. €6050-9570; B. €18480-26070; C. €12870-21560.	2004	Liljestam and Söderqvist (2004)

		and nursery for many aquatic species; refugia for birds; biodiversity maintenance.			based on three scenarios: A. in land, or 5 km offshore; B. in land, or 25 km offshore; C. 5km offshore, or 25 km offshore.			
Food provision (fish); amenity and recreation (biodiversity conservation or improvement, bathing and fishing improvements)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	Sweden (south west coast)	Europe	Estimation of WTP to improve recreational possibilities (bathing, fishing, biodiversity) via water quality improvements.	Estimated MWTP in € (in 2007)/year: €158 (biodiversity conservation); €68 (biodiversity improvement); €68 (improved bathing water quality); €147 (improved cod stock).	2002	(;) Eggert and Olsson (2003)
Food provision (fish); amenity and recreation (recreational fishing)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	Sweden (Kattegatt-Skagerrak area)	Europe	Benefit transfer (from Eggert and Olsson, 2004) to estimate a WTP for a cod-moratorium leading to an increase in the cod stock to the 1974 level in the Kattegatt-Skagerrak area.	WTP estimate: €27-152 (million) in 2007.	2004	(;) Franzén et al. (2006)

Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Sweden (Skagerrak area)	Europe	Benefit transfer (based on Hökby and Söderqvist, 2003) to estimate a WTP for a reduction in nitrogen loads in the Skagerrak area.	Annual WTP € in 2007: 77MEUR (for 10000tons/year reduction); 117 MEUR (for 25000 tons reduction); 162 MEUR (for 50000 tons reduction); 223 MEUR (100000 tons reduction); 234 MEUR (for 110000 tons reduction).	2003	Franzén et al. (2006)
Water quality (reducing eutrophication)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality.	Sweden (Baltic Sea)	Europe	Contingent valuation study to reduce eutrophication in the Baltic Sea (improving water quality).	Mean WTP € (in 2007): €197/person. Aggregate value: €1261 (million).	1995	Markowska and Zylicz (1999)
Water quality (reducing eutrophication)	Coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality	The Baltic Sea (Sweden, Denmark)	Europe	Economic valuation of increased water quality reducing the causes of eutrophication in the Baltic Sea. Method used: contingent valuation (WTP). Mail survey conducted in Sweden (estimated values for Denmark).	Swedish mean annual WTP €446-798 (in 2007) per person; Danish mean annual WTP €512-915 (in 2007) per person.	1995	Turner et al. (1999)

Water quality (reducing eutrophication)	Coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality	The Baltic Sea	Europe	Economic valuation of nutrient (nitrogen and phosphorus) reductions in the Baltic Sea. Method used: contingent valuation (WTP). Mail survey conducted in Sweden (estimated values for Denmark).	WTP per person (estimates based on WTP elicited in a Polish contingent valuation study) (€/2007): €156.6 (€79.4 - zero WTP assumed for non respondents)		1995	Turner et al. (1999)
Raw materials (grassland/hay and reeds); bioremediation of waste	Coastal areas (flood plains and coastal wetlands)	Sediment and nutrient cycling; primary production; water cycling; climate mitigation; regulation of water flow and quality.	The Baltic Sea	Europe	Harvested value of grassland/hay and reeds. Replacement cost value of reducing nitrogen loads in the Baltic Sea.		Market prices valuation for grassland/hay: € (2007)/ha: €81.5-151.4; reeds €145.1-944.5. Replacement cost method for nitrogen sinks € (2007)/ha/year: 1390.5.	1994	Gren et al. (1995)
Amenity & recreation (recreational fishing)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	The Baltic Sea and The Baltic Sea basins (rivers and lakes) in Denmark, Finland, Iceland, Norway and Sweden.	Europe	WTP for recreational fishing in the catchment and the marine areas of the Baltic region, and actual expenses, in the same area, for recreational fishing. Results are not exclusive for marine and coastal areas.	WTP estimate to preserve the area as it is (€ in 2007): €298 (million)/year.	Annual expenses for recreational fishing (€ in 2007): €326 (million). Shares of these total expenses are not specified.	1999	Toivonen et al. (2000)

Energy (gas extraction); multiple services	Coastal wetlands, marine	Sediment and nutrient cycling; primary production; water cycling; sediments and nutrients retention/storage	The Netherlands	Europe	CBA assessing original industry study			Wetten et al. (1999); Schuijt (2003)
Food provision (fish); carbon storage; amenity and recreation; disturbance alleviation and prevention	Coastal wetlands	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	UK	Europe	Restoration of coastal wetlands with managed realignment as a measure of adaptation to climate change. CBA to value benefits involved with restoration. Choice experiment (WTP) to estimate the recreational benefit of saltmarshes. Investigation of the influence of the living distance to saltmarshes.	Estimated MWTP: fisheries provision - £7.43 (conservative estimate) carbon storage - damage cost avoided - 4 different values between the range £7-230; amenity & recreation (including biodiversity) - choice experiment - £4.31 (use values) and £3.57 (non-use values)	2006	Luisetti et al. (2008a); Luisetti et al. (2008b); Luisetti et al. (2008c)
Food provision; raw materials; carbon storage; sea beds and coral banks (impacts); amenity and recreation	Marine	Sediment and nutrient cycling; primary production; water cycling; climate mitigation; regulation of water flow and quality; habitat and nursery for many aquatic	UK (Scotland - Atlantic frontier)	Europe	Identification of ecosystem goods and services at specific locations based on the ecosystem approach of the Millennium Ecosystem Assessment (UNEP, 2006) to validate a theoretical		Various years	Beaumont et al. (2007)

		species; biodiversity maintenance			Ecosystem Approach framework.			
Food provision; raw materials; carbon storage; amenity and recreation; resilience and resistance	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; climate mitigation; regulation of water flow and quality; habitat and nursery for many aquatic species; biodiversity maintenance	UK (Isles of Scilly)	Europe	Identification of ecosystem goods and services at specific locations based on the ecosystem approach of the Millennium Ecosystem Assessment (UNEP, 2006) to validate a theoretical Ecosystem Approach framework.	Tourism: 85% of the isles of Scilly economy.	Various years	Beaumont et al. (2007)
Food provision; raw materials; carbon storage; bioremediation of waste; cultural heritage; amenity and recreation	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; climate mitigation; regulation of water flow and quality; habitat and nursery for many aquatic species;	UK (north-east coast of England - Flamborough Head)	Europe	Identification of ecosystem goods and services at specific locations based on the ecosystem approach of the Millennium Ecosystem Assessment (UNEP, 2006) to validate a theoretical Ecosystem		Various years	Beaumont et al. (2007)

		biodiversity maintenance			Approach framework.			
Amenity & recreation (water quality - bathing/health impacts)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance.	UK (south-west of Scotland)	Europe	Revealed-preferences study to estimate the annual welfare benefit of increasing water quality to the standard required by the 1976 EU Water Quality Directive.	Estimated benefit: €11.40/person/year.	2001	Hanley et al. (2001)
Amenity & recreation (water quality - bathing/health impacts)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic	UK (south -west of Scotland : Ayr and Irvine)	Europe	Contingent valuation study to estimate the annual WTP of increasing water quality to the standard required by the 1976 EU Water Quality Directive for the towns of Ayr and	WTP € (in 2002)/person/year: €19.44 (Ayr); €11.84 (Irvine).	2002	Hanley and Kriström (2002)

		species; refugia for birds; biodiversity maintenance.			Irvine (Scotland).				
Food provision; raw materials; nutrient cycling; gas and climate regulation; disturbance prevention and alleviation; cognitive values; amenity and recreation	Marine and coastal areas (Marine Conservation Zones - MCZs)	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	UK	Europe	Benefit transfer method used to value the benefits provided by the Marine Conservation Zones (MCZs) in the case of three different configurations and two different management regimes.	Benefit values range £10.2-23.5 (billion) in present value terms using a 3.5 % discount rate. Values for each ecosystem service (form other studies): food provision £885 million; raw materials £117 million; nutrient cycling £1.3 billion; gas and climate regulation £8.2 billion; disturbance prevention and alleviation £440 million; cognitive values £453 million; amenity and recreation £1.4-3.4 billion.		(z) Hussain et al. (2009)	
Food provision (fisheries);raw materials; gas and climate regulation; disturbance alleviation and prevention; cognitive values; leisure and recreation; bequest and existence values	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	UK	Europe	Literature review on good and service valuations provided by UK marine ecosystems.	Bequest and existence values (non-use values): WTP for surviving species of sea mammals £19- 46/household/annual per species (depending on the species); total non-use value £469- 1,136 million (in 2004).	Fishery industry (food provision): £513 million (Defra fisheries statistics, 2004). Seaweed (raw materials) total gross income: £349,819-583,032 (European Parliament Report, 2004). Gas and climate regulation: £420 million - 8.7 billion (UK£2004) (Beaumont et al., 2006) Saltmarshes (disturbance alleviation and prevention): £17-32 billion (King and Lester, 1995; Beaumont et al., 2006). Cognitive values: UK marine research and	Various years	Nunes et al. (2009)

application: £292 million (in 2002); education and training in marine science: £24.8 million (Pugh and Skinner, 2002). Leisure and recreation: total net value UK £11.77 billion (Pugh and Skinner, 2002); Scottish whale-tourism total income £7.8 million (in 1994) (Beaumont et al., 2006).

Non European case studies

Food provision (customary fishing rights - bequest (non-use) value)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	Muaivuso peninsula	Fiji (Pacific Ocean)	Contingent valuation method used to assess the WTP (in money and/or time) to conserve the Navakavu iqoloqoli (customary fishing rights), a bequest value.	WTP for bequest values (non-use values): FJ\$1.25-1.41 (US\$ 0.69-0.73)/person/week or FJ\$ 183.90 (US\$ 106.91) per household per year.	2006	O' Garra (2009)
Recreation (tourism); food provision (fishing); raw material (seaweed farming; mangrove harvesting)	Marine and coastal areas	Sediment and nutrient cycling; primary production; water cycling; regulation of water flow and quality; habitat and nursery for many aquatic species; refugia for birds; biodiversity maintenance	Zanzibar (Tanzania)	Africa	Surveys and benefit transfer data.	Tourists' expenditure: US\$184.9 (million). Average annual fisherman earning: US\$765. Average annual seaweed production earnings: less than US\$58. Contribution to GDP from mangrove harvesting US\$28 (thousand).	2007	Lange and Jiddawi (2009)

Annex 4: Fisheries and Shipping services assessment case study

Introduction

Marine fisheries generate economic revenues, support livelihoods and provide food for human and cultured animals, and is a major ecosystem service provided by the ocean. In this section, fisheries refer to marine capture fisheries only, and do not include aquaculture. Financially, the gross revenue generated directly by global capture fisheries was estimated to be about \$US 85 billion in 2004 value (The World Bank and FAO, 2008). Moreover, the fishing sectors receive substantial subsidies from government that amounts to as much as US\$ 26 billion globally (Sumaila and Pauly, 2006). It is estimated that the total net profit of the global fisheries is negative and in the order of \$US 5 billion in 2004 value (The World Bank and FAO, 2008). The negative net profit is mainly caused by the loss of fisheries productivity resulted from over-exploitation of many fisheries resources in the world. However, marine fish and shellfish remain as an important source of animal protein. Fish contributes to 15.3% of world's total animal protein intake in 2005 and the corresponding figure in Europe is about 11% (FAO, 2008). In coastal Low-Income Food-Deficient Countries (LIFDCs), fish contributes at least 20% of animal protein intake (FAO, 2008; Swartz and Pauly, 2008). Global per capita fish consumption has been increasing steadily in the past four decades and the per capita fish supply in Europe is 20.8kg/year in 2005 (FAO, 2008). Fishing is the major, and in many cases, the only available livelihood for many coastal communities.

Globally, majority of exploited fisheries resources are fully- or over- exploited. Global reported fish catch peaked at around 80 million tonnes in the mid-1980s (FAO, 2008). Up to 10% of the global catch is contributed by the highly productive and variable Peruvian anchovy (*Engraulis ringens*). If catch from Peruvian anchovy is excluded, global catch shows a steady decline since the 1980s. Overfishing is a major reason for the observed decline in catch of many fish stocks, FAO (2008) estimated that of all the marine fish stocks reported in the catch statistics, 19% are overexploited, 8% are depleted, 52% are fully exploited, 20% are moderately exploited, and only 1% demonstrated signs of recovery from overexploitation. A recent study shows that 63% of the fish stocks worldwide that are examined in the study require rebuilding and reduction in exploitation rates, in order to recover those stocks (Worm *et al.*, 2009).

Fishing activities exert certain impacts to fish populations and marine ecosystems which may have negative effects on the goods and services (including fisheries itself) provided by the marine environment. The scale and level of such impacts depend on the intensity of fishing activities and the availability and effectiveness of management measures. Excessive and irresponsible fishing may deplete exploited fish populations. At the extreme case, fishing may drive exploited populations to local extinction. This applies to both target and non-target species that are affected through by-catch and

other incidental mortalities resulted from fishing activities (e.g., mortality from discard fishing nets). The modification of fish population dynamics may lead to ecosystem-level changes, such as the release of prey populations as predators are depleted by fishing. Moreover, some unsustainable or irresponsible fishing causes excessive damage to habitats, for example, damage of benthic structural habitats such as coral or sponges beds by bottom trawling. Such negative impacts of fishing reduce the productivity of fish stocks, affecting biodiversity and the functioning of marine ecosystems.

In addition, fishing activities may have other environmental cost. Industrial fishing is fossil fuel intensive. Fisheries burned almost 50 billion L of fuel annually in the process of landing just over 80 million tonnes of marine fish and invertebrates for an average rate of 620 L t⁻¹ (Tyedmers *et al.* 2005) As a result, it directly emits more than 130 million t of CO₂ into the atmosphere, contributing to global anthropogenic climate change.

In this section, we aim to discuss the approaches to investigate socio-economic values of North East Atlantic marine ecosystem in the context of commercial fisheries. Ideally, a Total Economic Valuation (TEV) should account for all the above economic costs and benefits. However, this section will focus on financial valuations of commercial fisheries of OSPAR countries in North East Atlantic as an indication of their economic values. We use the Driver Pressure State Impact Response (DPSIR) framework to assess fisheries of OSPAR countries. We then discuss economic analyses such as cost-benefit analysis (CBA) and economic-effectiveness assessment (CEA) to predict the gain and loss of human welfare under different management and policy options. Four scenarios are adopted in this analysis and they are market globalization, global sustainability, provisional enterprise and local stewardship (Turner, 2005). The results will be useful for decision makers to design and implement management policies with highly unavoidable uncertainty in the future.

Driver Pressure State Impact Response (DPSIR) framework

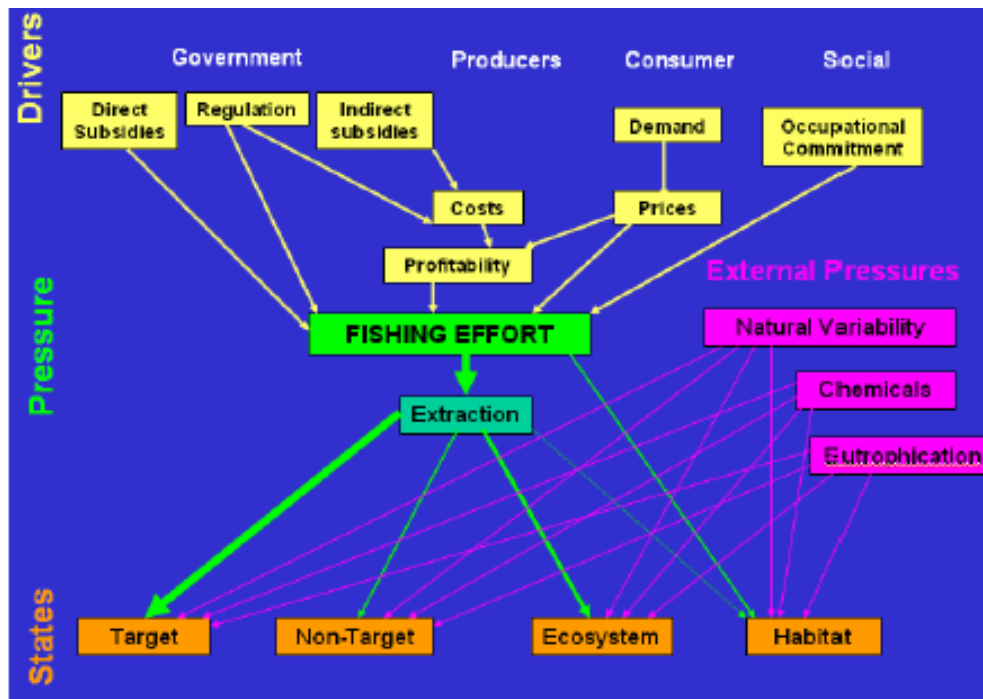
One of the tools for scoping sustainable development issue is the Driver Pressure State Impact Response (DPSIR) framework, which was first used by the OECD and further developed and adapted by Turner (1998) for coastal zone management. An European Lifestyles and Marine Ecosystems (ELME) project applied the DPSIR framework to Basque trawlers operating in North East Atlantic (Hoff *et al.*, 2008). Figure 13 summarized the cause and effect relationship using Drivers, Pressures and States indicators. In the case of multi-species fisheries, the main drivers include increasing consumer demand on seafood, highly dependent on fisheries for jobs, subsidies from governments to the fishers and the value added by fisheries to the Gross Domestic Product (GDP). These will then create pressures which include boosting up the fishing effort, fleet size, investment and eventually the fish catches. These pressures exert an impact on fish populations and marine ecosystems, catch per unit effort and landed value of fishes. These changes will then have an impact on human benefits, for example, impact on net

revenue from capture fisheries and jobs that are dependent on fisheries. These impacts will induce the change in management options and these changes will control the socio-economic drivers, pressures, state and impacts. Thus, this framework is a dynamic cycle with feedback processes (Hoff *et al.*, 2008; Ledoux and Turner, 2002). The DPSIR framework used in this analysis has been summarized in Figure 14.

Economic indicators

Indicators are necessary for valuing the ecosystem services. FAO (1999) and the EU fisheries data collection programme (Commission Decision, 2008) identified various economic indicators for assessing sustainable development of capture fisheries and economic performance of European fishing fleets respectively (Hoff *et al.*, 2008). We selected a subset of these indicators for our DPSIR framework (Table 5A).

Figure 13 DPSIR model applied to trawlers in North East Atlantic



Source: European Lifestyles and Marine Ecosystems, by EU (EC/DG Research: Contract GOCE-CT-2003-505576).

Figure 14 DSPIR model applied to socio-economic analysis of fisheries in North East Atlantic

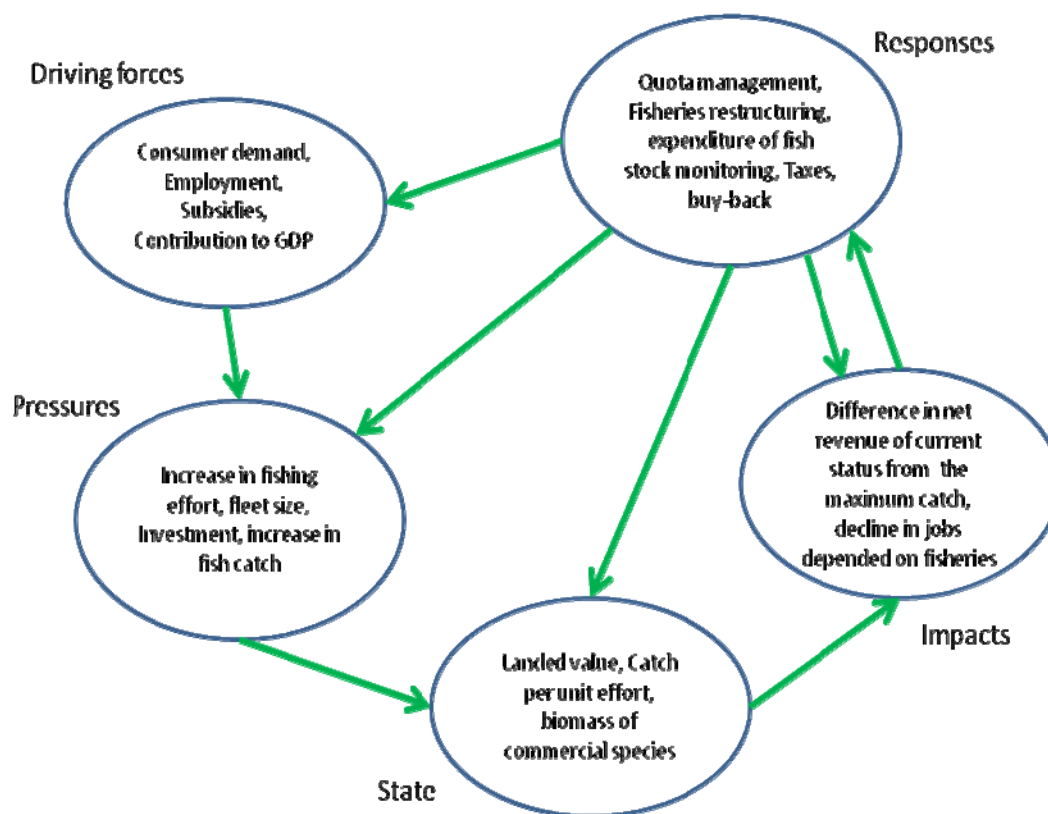


Table 5A Selection of indicators, unit of measurement, level of aggregation and data sources

Indicators	Variables	Unit of measurement	Level of aggregation	Data sources	Notes
Drivers					
Consumer Demand	Fish consumption per capita	kg	By group of fisheries commodities and processed products	FAO/FIDI (Fishery Information, Data and Statistics Unit) (1961-97)	Indicator of the demand of consumer on seafood
Employment in the fishery sector	Employment on board	Full-time Equivalent (FTE)	Total employment in fishery sector in each country	<ul style="list-style-type: none"> Eurostat New Cronos database, Agriculture and fisheries, EAUF FAO 	
Subsidies	Subsidies	USD/Euro	By sub-sectors/fleet/fishery	SAUP subsidies database	
Contribution to GDP	% of the total GDP	% of the total GDP	By country	Eurostat	% of the total GDP contributed by fishing sector in each country
	Per capita GDP	Euro per inhabitant	Annual GDP per inhabitant in each country	Eurostat	
Pressures					
Fishing effort	Vessels power	Horse power per year	By country, By fishery segment	International Council for the Exploration of the Sea (ICES): ACFM http://www.ices.dk/committe/acfm/acfm.htm SAUP	

	Days at Sea	Days	By country, by fleet	Economic performance of selected European fishing fleets', <i>Annual Report</i>	
	Energy consumption	Litres	By country, by fleet	Sea Around Us Project (SAUP) database	
Fleet size	Fleet Number	Number of vessels	By country, by type of vessel	Eurostat, Fisheries DG, Blue Plan	Excess capacity may lead to overfishing
	Fleet vessels tonnage	total tonnages	By country, by type of vessel	Eurostat, Fisheries DG, Blue Plan	
Investment	Investment in physical capital	Euro	By country, by fleet type	FAO, EAUF	
Fish catch	Landings	Tonnes	Annual catch by country, species or higher taxonomic level, FAO major fishing areas	FAO/FIDI: http://www.org/fi/struct/fidi.asp#FIDS FAO/FAOSTAT: http://apps.fao.org ICES http://www.ices.dk/fish/statlant.htm Eurostat: http://europa.eu.int/comm/eurostat <i>Statistics in focus</i> Blue Plan: http://planbleu.org	
State					
Income	Landed value	EUR/USD	By country, by species or higher taxonomic group, fishing fleet	'Economic performance of selected European fishing fleets', <i>Annual Report</i> , SAUP	
Catch	Catch per unit effort	Tonnes/horse power	Catch per unit of effort	ICES, FAO/FIDI, FAO/FAOSTAT, Eurostat	
Exploitation Status of fish stocks	Number of overfished stocks	Number of species		International council for the Exploitation of the Seas (ICES)	
Fish abundance	Biomass of commercial species	Tonnes	By species or higher taxonomic group	ICES (http://www.ices.dk/committe/acfm/acfm.htm)	Indicator of the state and size of spawning stock
Impacts					
Change in net revenue	Net revenue	EUR/USD	By country, by species or higher taxonomic group, fishing fleet	FAO, 'Economic performance of selected European fishing fleets', <i>Annual Report</i> , SAUP	Landing value – variable cost
Change in jobs	Employment in fishery sector	Number of employees	Total employment in fishery sector in each country	Eurostat New Cronos database, Agriculture and fisheries, Economic performance of selected European fishing fleets', <i>Annual Report</i>	
	Employment on board	Full-time Equivalent (FTE)	Total employment in fishery sector in each country	Economic performance of selected European fishing fleets', <i>Annual Report</i>	
Responses					
Quota management	Quota	Weight of fish (Tonnes)	TAC per area and season	Fisheries DG (http://europa.eu.int/comm/dgs/fisheries/index_en.htm)	
Fisheries restructuring	Numbers of vessels within specified	Number	By countries, fleet type	EAUF, FAO	Number and size of vessels is a

	categories (overall length, engine power or type of gear in use)				direct measure of fishing effort
Expenditure on fish stock monitoring	Expenditure on fish stock monitoring	Euro/USD, days at sea for research vessels, area covered by monitoring survey	By country	Published literature and local fisheries agencies.	Indicate how much money each country is invested to ensure management measures are followed.
Taxes	Taxes	EUR	By country		
Buy-back	Expenditure on buy-back	EUR	By Country		

From financial valuation to environmental impact assessment

A range of socio-economic drivers such as increasing consumer demand in seafood, reliance on fisheries for employment, increasing in financial aid (subsidies) from the governments exert certain extent of impact on fisheries resources and hence affect economic efficiency of the fishery sector. Other natural and anthropogenic drivers related to marine environment including climate change, eutrophication and oil spill incidents may also add to the impact of human activities on economic efficiency of fisheries. These impacts eventually cause the economic surplus (rent) from the fishery sector to increase or decrease. An appropriate approach for assessing the economic efficiency of different policy options under different scenarios is cost-benefit analysis (CBA). Cost-benefit analysis is a systematic economic analysis by evaluating the economic costs and benefits of the public policy. Alternatively, cost-effectiveness analysis (CEA) could be employed to compare the cost of alternative policies or approaches to achieve some predetermined targets or objectives e.g., biodiversity conservation and maintaining landings at Maximum Economic Yield (MEY).

Cost-benefit analysis

Cost-benefit analysis assesses the change in net social benefits which comprise of both consumer (consumer surplus) and producer (producer surplus) net benefits. In practice, when evaluating net benefit of fisheries using CBA, only change in producer surplus is considered and change in consumer surplus is ignored. One of the reasons for not considering consumer surplus is the presence of a number of substitutes in a competitive market, so the consumer prices are not depended on the change in supply of a particular marine species. That means the demand is assumed to be perfectly elastic, so the impact on consumer surplus can be ignored.

Also, CBA in socio-economic assessment involves economic analysis which focuses on the costs contributed and benefits gained by the society as a whole (in contrast, a purely financial analysis only focused on the costs and

benefits related to private organisations, firms and individuals). Some adjustments on financial values are necessary before assessing economic costs and values. First, transfer payments are excluded in the economic analysis. Transfer payments represent transfer of resources from one sector or member of society to another but do not represent direct extraction from country's marine resources. As such, transfer payments including taxes, interest payments and depreciation are not included in the cost-benefit analysis. The second adjustment is to include externalities which are the costs and benefits incurred to society but do not show up in the financial profit and loss of the fisheries sector. In the context of fisheries, external costs include the economic loss from discarding of by-catch marine species, non-target species, and other charismatic species like marine mammals and sea-birds. The third adjustment is to embrace all costs necessary for achieving the project's benefits into the cost-benefit analysis. For instances, the costs for implementing and enforcing a policy such as individual transferrable quota programme.

Since the decision rule generally used in CBA is the net present value (NPV) rule, the financial values of the indicators discussed above are assessed in terms of net present value. The net present value is given by

$$NPV = \sum_{t=0}^T B/(1+r)^t - \sum_{t=0}^T C(1+r)^t - \sum_{t=0}^T EC(1+r)^t$$

where NPV = net present value, $t = 0$ is the current time, T is the terminal time of the analysis, B is the benefits in time t , C is the operating costs in time t , EC is the external cost in time t , and r is the discount rate. In this analysis, only variable (operating) costs are considered, whereas sunk costs are excluded. Sunk costs are defined as the costs that have been incurred before the policy is implemented, for example, capital cost of vessels and fishing gears. The choice of discount rate is also critical and a standard discount rate is necessary for different projects of OSPAR countries, so the costs and benefits of projects can be compared. Thus, OSPAR countries must get a consensus on a single discount rate for their projects. However, it is not possible to have a single discount rate for satisfying all the criteria of commercial viability, environmental sustainability, and social responsibility. So, several discount rates can be selected for performing sensitivity analysis which shows how sensitive the results are to the choice of discount rate.

The values of these indicators under the current situation are first estimated from the sources that described in Table 5A and/or other grey literatures. The change in the gross landed values, net profit, jobs associated with fisheries and other indicators will be estimated under different scenarios with different policy options. Scenarios analysis allows us to deal with uncertainty over temporal scale. In performing the CBA for fisheries, the net present benefits from fisheries without the policy are compared against the NPV with the policy under different scenarios. Future catch per unit effort, biomass and landed values can be predicted by adjusting the effort level and productivity using numerical models under different policy options and scenarios. For simplicity, ex-vessel prices of commercial marine species can be assumed to be

constant throughout the temporal scale. This may be a reasonable approximation as the integration of global fish markets stabilizes fish demand. This latter may be the reason for the apparent lack of noticeable increases in the real price of fish in general recently (Sumaila et al., 2006). Alternatively, a fish price model that accounts for change in demand and supply couple be incorporated.

In the past, CBA has already demonstrated as a successful tool for identifying the best policy option for managing marine resources. Here are some of the examples of CBA application on fisheries:

Box A4-1

Examples of CBA applied on fisheries:

Freese, S., Glock, J. And Squires, D. 1995. Direct allocation of resources and cost-benefit analysis in fisheries – an application to Pacific whiting. *Marine Policy*. 19 (3) 199 – 211.

Cost-benefit analysis for comparing economic efficiency for allocating Pacific whiting among three different policy alternatives.

Herrick Jr., S. F., Strand, I., Squires, D., Miller, M., Lipton, D., Walden, J. and Freese, S. 1994. Application of benefit-cost analysis to fisheries allocation decisions: the case of Alaska Walleye Pollock and Pacific Cod. *North America Journal of Fisheries Managment*. 14: 726 – 741.

Reviewed the proposals for allocating Alaska Walleye and Pacific Cod using benefit-cost analysis in U.S. fisheries off Alaska. Conceptual and practical problems associated with benefit-cost analysis are discussed as well as their solutions.

Schwindt, R., Vining, A. And Globerman, S. 2000. Net loss: a cost-benefit analysis of the Canadian Pacific Salmon fishery. *Journal of policy analysis and management*. 19(1): 23 – 45.

The cost-benefit analysis of the Canadian Pacific Salmon fishery demonstrated that the government policies to preserve the fishery have a higher social costs than that resulted from a “do nothing” policy.

Cost-effectiveness analysis

To find out the best policy to achieve some policy goals, an alternative approach is to use the cost-effectiveness analysis (CEA). In this approach, we aim to determine the most efficient policies to achieve a given policy goal. This is particularly applicable when the information about the benefits of altering environmental quality, fishers' behaviour and government policies on marine resources is scarce. Then, the decision rule would be to select the policy with the lowest sum of present value of costs. In the context of fisheries, the policy goal can be set to achieve a particular biomass level, for example,

the maximum sustainable yield (MSY) and maximum economic yield (MEY), of a particular species.

Conclusion

This section described the elements (indicators) and framework (DSPIR) of applying economic analysis on fisheries. The framework and analysis aim to aid policy-makers to decide which policies are the best for managing marine resources. The cost-benefit analysis may be useful for allocating total allowable catch (TAC) and national quotas among nations of OSPAR countries. However, these economic analyses are criticized for not being able to capture the full value of potential losses, for example, the losses of species and the social impact of coastal communities, because of the difficulties in capturing these values in economic terms. To have a more comprehensive economic analysis on fisheries, the total economic values (TEV) is recommended to reflect the full scope of costs and benefits faced by the society under different policy options. Even in cases where the full spectrum of economic values could not be captured, economic analysis (CBA and CEA) using adjusted financial costs and benefits are still valuable. Results from these analyses allow us to compare and contrast the benefits or efficiency of different policy options.

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