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Work Package 1:

Managing Marine Ecosystem Services:

A review of the Ecosystem Approach

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Summary

This report reviews the current state of the science and best practice relating to the implementation of the *Ecosystem Approach* (EA) to managing ecosystem services in coastal waters and shelf seas. The report focuses on fisheries and marine ecosystems. Much of the debate about the failure to manage marine resources has focused on governance and socio-economic matters but a detailed review of these topics was beyond the scope of this report.

An ecosystem is defined as any area of nature that includes living organisms and non-living substances interacting to result in an exchange of materials between the living and non-living parts. Marine ecosystems are composed of organisms that range in size from microscopic plants (phytoplankton) and animals (zooplankton) to fish and mammals. Collectively, these organisms make up the food web, a complex and variable network of feeding linkages and feed-back loops.

Humans have been using the services provided by marine ecosystems for thousands of years but in doing so put these ecosystems under pressure through the release of contaminants, enrichment of waters with inorganic nutrients, the requirements for renewable energy, and the generation of noise and litter. Commercial fishing has a large impact on our marine ecosystems as it removes large top predator fish species, which often results in by-catch of non-target species associated with a variety of fishing methods and fishing activity that can cause disturbance/ destruction of the benthos.

Individually and collectively, these pressures have been shown to influence the structure and functioning of marine ecosystems and in turn the services they provide. Furthermore, the interactions between different pressures can result in unexpected impacts on ecosystems and it is evident that changes in ecosystem state brought about by one pressure can negatively affect other services. Managing human activities to minimise the impact on ecosystem structure and function is not a trivial task. In addition to human driven changes, marine ecosystems are dynamic and subject to natural variability over a range of spatial and temporal scales and are being influenced

by long-term climate change (global warming). Some marine ecosystems have undergone an abrupt regime shift. This represents a particular challenge in managing ecosystem services because the new regime will be different from what occurred before, it is likely to persist and the services it provides may differ from those previously available.

The strategies used to manage the influence of human activities on marine ecosystems are viewed as having been largely unsuccessful. This view together with an awareness of the growing pressure on these systems, has led to the development of ecosystem based strategies that recognise the intrinsic value of marine ecosystems and that humans are part the ecosystem. The ecosystem approach is an integrated (or holistic) strategy, rather than the more typical sector by sector approach and there is an expectation that the EA will: maintain ecosystem structure and functioning, thereby ensuring the sustainable use of resources; avoid negative interactions between different human activities; reduce the risk of failing to identify the cumulative effects of different pressures.

Much of the debate about the ecosystem approach has focussed on fisheries and what has been referred to as ecosystem-based fishery management (EBFM). There has been less discussion of using the EA concept to manage the collective pressures arising from the human use of ecosystem services. However, the elements that underpin EBFM (such as giving greater consideration to the ecosystem and not focussing solely on the resource) appear much the same as for other pressures. In our opinion, focussing on fisheries risks perpetuating the much criticized sectoral approach and is inconsistent with the expectation of the integrated management of marine ecosystem services.

The North Pacific ground fish fisheries are an example of how the ecosystem approach has been applied to fisheries management. The objective was to provide future generations with the opportunities and resources "we enjoy today". To achieve this four goals were identified: (i) to maintain biodiversity consistent with natural evolutionary and ecological processes, including dynamic change and variability; (ii) maintain and restore habitats essential for fish and their prey; (iii) maintain system sustainability and sustainable yields of resources for human consumption and non-extractive uses; (iv) maintain the concept that humans are components of the ecosystem. Management includes: using maximum sustainable yield (MSY) as a limit not a target; setting limits on fishing in important wildlife habitat areas; restricted fishing on fish that are food for conservation species; setting reference points for non-target species; establishing marine protected areas (MPAs).

There is growing interest in the use of reference points for non-target species and MPAs as tools for reducing the impact of human use of marine ecosystems. Reference points can take the form of rearing success (chicks fledged per pair) as in the case of North Sea kittiwakes (*Rissa tridactyla)*, or as a limit on by-catch such as the OSPAR ecological quality objective for North Sea harbour porpoise *(Phocoena phocoena*). Marine protected areas with no-take reserves combined with limited effort in remaining fishing areas can be an important means of rebuilding depleted stocks. Furthermore, zoning coastal waters and shelf seas into un-fished marine reserves and areas with limited levels of fishing effort has been suggested as a way of ensuring that fisheries are part of functionally diverse ecosystems and therefore sustainable.

Reference points for non-target species and MPAs are likely to be an important component of the ecosystem approach. Nevertheless, their use requires careful consideration. Establishing MPAs and non-target species reference points to 'restore' the ecosystem may not achieve the desired goal because recovery back to a notional 'pristine' or 'background state' may not occur even when pressures are removed. Furthermore, it should not be assumed that top predator fish and mammal populations reflect the state of the ecosystem as a whole, or that conditions within an MPA can be used as reference conditions.

In a European context, a key challenge it to merge current and evolving methods of setting levels of harvest with measures to minimise the impact of fisheries on ecosystem structure and function. The European Commission considers MSY (the maximum that may be taken year after year) to be a key element of the Common

Fisheries Policy (CFP) and that by 2015, European community fisheries will be managed using MSY. Future fisheries advice from the International Council for the Exploration of the Sea (ICES) will be based on an integration of the precautionary principle, MSY and the ecosystem approach. Consideration of ecosystem state is not currently used in advice to set levels of harvest but ICES has recognised that restrictions on yields may be required to achieve Good Environmental Status (GES) for the Marine Strategy Framework Directive.

The ecosystem approach raises important questions about how to assess ecosystem state and quantify changes in state and how to use this information to manage the pressures associated with human use of ecosystem services. Some advocate the use of a suite of ecological indices. The establishment of operational objectives with indicators and reference points (values associated with a particular state of the ecosystem) is part of the ICES guidance on applying the ecosystem approach. Ecological quality objectives (EcoQOs) are under development for the North Sea by OSPAR but these relate to particular components of the ecosystem and it is unclear how they can be used for whole ecosystem properties such as biodiversity. An alternative is to use emergent ecosystem properties such as resilience (related to structure and function) that maintains and restores the integrity of an ecosystem in response to disturbance. However, further work is required to gain a better theoretical understanding of emergent ecosystem properties and how such properties can be used in management.

At the present time, the best approach is to use indicators and reference points for particular ecosystem components because for some the pressure – impact relationships are well understood and thresholds or targets have been or can be established and used to underpin management action. Nevertheless, it is likely that the disturbance of one or more ecosystem components can propagate into other parts and considerable caution is required to make sure that monitoring is not undertaken in a piecemeal fashion. In our opinion, it is important to recognise that such a focus may fail to identify important changes in whole ecosystem properties and the likelihood of regime shift.

The report makes three recommendations: (i) Fisheries management should be undertaken as part of a holistic approach to managing the human use of ecosystem services; (ii) the Department should undertake an assessment of the Irish Sea to determine: current ecological state; quantify the probability of future change in state; determine the economic consequences of any future change in ecosystem state on the sustainable use of ecosystem services; (iii) undertake further research on emergent ecosystem properties and their use in providing a holistic assessment of ecosystem structure and functioning.

1. Introduction

Coastal and shelf sea ecosystems provide humans with food (fisheries and aquaculture) energy, assimilate some of our waste and are a source of recreation and leisure. We have been using these '*ecosystem goods and services*' or more simply '*ecosystem services*' for thousands of years but often in ways that have been unsustainable. According to Lotze (2007) history is full of examples of the decline or collapse of a marine resource. The oceans cover 71% of the earth's surface and marine resources were once viewed as limitless. However, less than 7% of the 363 million square kilometres of the surface area of the oceans is shelf water < 200m deep and yet this supports 90% of the global fish catch (UNEP, 2008). Dredging of the sea bed has caused loss of important habitat. Over-exploitation has caused the collapse of some fisheries (e.g. Bundy & Fanning, 2005) and according to Jennings (2007) fisheries are viewed as an example of a failure in natural resource management. Exceeding the capacity of our waters to assimilate inorganic nutrients from domestic and agricultural sources has led to coastal eutrophication (Gowen et al., 2008) and the associated undesirable consequences are considered a major threat to coastal marine ecosystems (Nixon, 1995). The need for renewable energy such as that generated by offshore wind parks require space in coastal waters and this is considered to have the potential to influence the habitats of marine organisms (Hiscock et al., 2002). Finally, as discussed later in this report, there is the potential for the effects of one human pressure to have a negative influence on other services humans derive from marine ecosystems and for the impact of one pressure to exacerbate the impact of a different human pressure.

There is a general perception that strategies to manage the human pressure associated with the use of particular ecosystem services have met with limited success (but see Hilborn, 2007). Reasons for this include managing resources on a sector by sector basis (Sherman & Duda, 1999), failure to recognise the importance of marine ecosystems (Lotze & Milewski, 2004) and failure of governance (Browman & Stergiou, 2004). The aspiration to achieve better management of marine ecosystem services is evident at national and transnational/ international levels. For example, the vision for the UK Marine Science Strategy (Defra, 2010) is for: "*Clean, healthy, safe, productive and biologically diverse oceans and seas*" and the European Union Marine Strategy Frame Work Directive (OJEU, 2008) states that:

> "*The marine environment is a precious heritage that must be protected, preserved and, where practicable, restored with the ultimate aim of maintaining biodiversity and providing diverse and dynamic oceans and seas which are clean, healthy and productive, ensuring that the collective pressure of such activities is kept within levels compatible with the achievement of Good Environmental Status*..."

The concept of the ecosystem approach is not new (Smith, 1994) but is one that has received much attention as providing the means to better manage human pressures on marine ecosystems thereby protecting the structure and functioning of these systems and ensuring the long-term sustainability of the services they provide (Mace, 1997).

This report reviews the ecosystem approach in the context of marine fisheries and ecosystems but it is important to note that a key aspect of the ecosystem approach is that it is an integrated management approach, taking account of all human pressures of which fishing is only one. Furthermore, the successful management of marine resources include matters related to governance, economics and civil society as well as consideration of the structure and functioning of marine ecosystems. However, these topics were outside the scope of this report.

2. Marine Ecosystems and Human Influence

2.1 Introduction

The term ecosystem is defined as any area of nature that includes living organisms and non-living substances interacting to result in an exchange of materials between the living and non-living parts (Odum, 1959). A characteristic of marine ecosystems is that they are composed of organisms that range in size from microscopic plants (phytoplankton) and animals (zooplankton) to fish and mammals such as dolphins and whales. Collectively, these organisms make up complex and variable networks of feeding (trophic) linkages and feed-back loops all together called *food webs* (Figure 1).

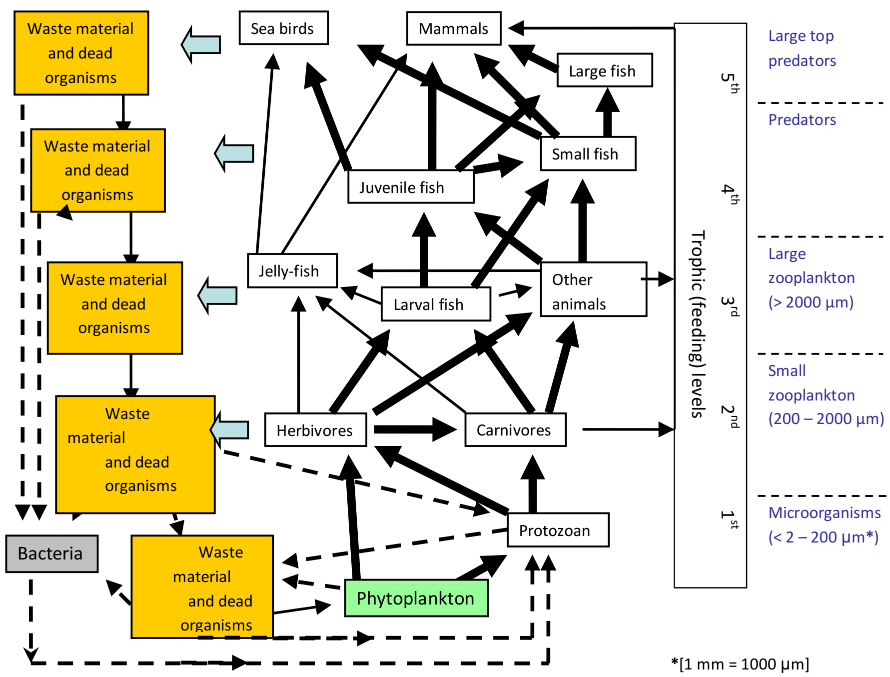

Figure 1. A marine pelagic food web.

Beyond shallow inshore waters, the microscopic floating plants that are collectively called phytoplankton form the base of the marine food web. Like plants on land, members of the phytoplankton use the energy from sunlight to make organic matter from carbon dioxide, water and nutrients such as the mineral salts of nitrogen and phosphorus. All of the animals including sea-birds, mammals, fish and those animals that live on and in the sea bed depend on phytoplankton production.

Many of the organisms that live in the sea are '*planktonic*' (drifting or free floating) or have planktonic larval stages. This means that the life cycle of these organisms is intimately linked to the physical environment. In the western Irish Sea for example: the seasonal production of phytoplankton only begins when solar warming of the sea surface creates a shallow surface mixed layer (Gowen et al., 1995); the distribution and abundance of larval fish is coupled to the seasonal pattern of water movement

(Dickey-Collas et al., 1996); the life history of Dublin Bay prawn (*Nephrops norvegicus)* is closely coupled to the physical environment of the western Irish Sea and the phytoplankton production season (Hill, 2007). The key point here is that changes in the plankton and planktonic life stages of (benthic) animals caused by perturbations of the physical environment may be propagated through the food web and influence higher trophic levels (including commercially fished species).

The diversity and abundance of species in marine ecosystems are not fixed in time and space. In addition to the characteristic seasonal cycle of growth in temperate seas, populations of plankton exhibit natural variability that ranges from short-term (days to weeks) to inter-annual variation in response to local meteorological variability. Decadal scale, variation in climate has been shown to influence the abundance of plankton (Edwards et al., 2006) and there is an expectation that the geographical distribution of some organisms will be influenced by long-term (~100 years) global warming. One example is in change in the geographical distribution of the copepods *Calanus finmarchicus* and *C. helgolandicus* (important food for cod, *Gadus morhua*) reported by Planque and Frédou (1999). Populations of fishes are not constant in time and space and are also subject to climate variation. It has been suggested, for example, that the southern range of cod may be reduced if bottom water temperature exceeds 12° C (Drinkwater 2003, ICES 2008a). Marine ecosystems may also undergo periodic and abrupt 'regime shifts': '*relatively sudden changes between contrasting, persistent states of a system*' (De Young et al., 2008); '*sudden changes in ecosystem structure that can be detected across several ecosystem components*' (Spencer et al., 2010); "*abrupt changes in the structure and functioning of the ecosystem over a large area*" (Lees et al., 2006). The changes that took place in the biomass of individual species at several trophic levels (including horse mackerel (*Trachurus trachurus*)) in the North Sea in the late 1980s (Reid et al., 2001) provide one example. Regime shifts represent a significant challenge in managing ecosystem services. This is because the new regime may be different from what occurred before, is likely to persist, and the services it provides may not be the same and those previously available. It is therefore important that the adoption of the ecosystem approach takes account of natural (seasonal, inter-annual and decadal scale) variability and changes that may arise due to long-term climate change (global warming) and the potential for regime shift.

2.2 Human impacts on coastal and shelf sea ecosystems

2.2.1 Eutrophication

One of the main human pressures that coastal marine ecosystems are facing is nutrient enrichment. This can lead to eutrophication[12], which is seen as a major threat to the functioning of coastal ecosystems and a worldwide problem of increasing severity (McIntyre, 1995). Generally, under conditions in which phytoplankton growth is constrained by the availability of nutrients rather than light, nutrient input can enhance the growth of primary producers. Many coastal areas of northern Europe are enriched with inorganic nitrogen and phosphorus and a number of studies have related increased phytoplankton biomass and production to enrichment over the last 30-40 years (Radach et al., 1990; Schaub & Gieskes, 1991; De Jonge et al., 1997; Gowen et al., 2000). Some increased production might be viewed as good if the additional organic matter is transferred to higher trophic levels because it could mean that more fish are available to be harvested. However, under certain conditions increased phytoplankton (and macro-algal) production and elevated biomass can have a negative impact on marine ecosystems and the services they provide. It is not so much the size of the nutrient input, but rather the uncoupling between production and its transfer through the food web that can cause problems (Tett et al., 2007). The negative effects of increased phytoplankton biomass and production have been summarised by Gowen et al. (2012) and include an increase in the occurrence of Harmful Algal Blooms (HABs), deep water de-oxygenation (as the extra organic matter is decomposed) and

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[2] Eutrophication is defined by the EU Urban Waste Water Treatment Directive (CEC, 1991) as: "*the enrichment of waters by nutrients, especially compounds of nitrogen and/or phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned*." (See also Ferreira et al., 2011).

associated benthic mortalities, fish kills and the loss of aquaculture production. More subtle changes include shifts in species composition and community structure (Officer & Ryther, 1980; Gillbricht, 1988; Gifford & Caron, 2000; Cadèe & Hegeman, 2002; Lancelot et al., 2006), that could, in turn alter food web structure and nutrient cycling.

Many countries have introduced legislation to manage the nutrient discharges to coastal waters to prevent or minimise the potential undesirable ecological and socio-economic (including effects on human well-being and health, Nixon, 1995) consequence of eutrophication. Within the European Union, the Urban Waste Water Treatment Directive (CEC, 91/271/EEC) and Nitrates Directive (CEC, 91/676/EEC) were established for this purpose and nutrient removal from discharges is required if a water body is diagnosed as eutrophic or at risk of becoming eutrophic. The Oslo Paris Convention (for the protection of the waters of the North East Atlantic) strategy to combat eutrophication (OSPAR, 2003) serves a similar purpose. Eutrophication is also one of the eleven descriptors by which the state of water bodies will be assessed under the requirements of the Marine Strategy Framework Directive (OJEU, 2008).

2.2.2 The effects of commercial fishing on ecosystems

The ecological consequences of commercial fishing have been well documented and include the effects of removing top predators, the impact of trawling on the benthos and removal of fish that are prey for marine organisms. Fisheries tend to remove large, slower-growing fish species first and these are typically the species that occupy higher trophic levels (TL) (see Figure 1). For example, species such as cod and tuna that feed on other fish, occupy a higher trophic level in the food web. In comparison, species such as sardines that feed lower down the food chain (on zooplankton) occupy a trophic level of ~3.0 (Froese & Pauly, 2000). Removal of top (higher trophic level) species can therefore result in the mean TL of the remaining fish species being lower (Pauly et al., 2002). The net result is the equivalent of '*fishing lower down the food chain*' and an apparent decline in mean trophic level. A declining TL can also occur within a fish population as fishing pressure reduces the age spectrum of the population and smaller adults feed lower down the food chain.

Pauly et al. (2002) stated that the observed decline of 0.05 – 0.10 trophic levels in global fisheries landings per decade since the 1970s, was severe and implied the gradual removal of large, long-lived fishes from the world's oceans. However, this analysis assumes that the global increase in fisheries at lower TL is solely driven by the lack of fishing opportunities for top predators and this is not necessarily the case. According to Pauly et al. (2002) one consequence of the removal of top fish predators was a reduction in the 'buffering' (i.e. resilience, see section 4) capacity of marine food webs. Jackson et al. (2001) made a similar point and further suggested that the removal of large fish species by fishing has resulted in the '*microbialization*[3]' of coastal food webs.

The removal of top predators can influence ecosystem structure and function by causing what have been termed 'trophic cascade' and regime shift. Here we summarise three examples. The first relates to the fishing of sharks on the eastern seaboard of the USA between Cape Cod and Cape Canaveral. Myers et al. (2007) investigated the cascading effects resulting from the loss of this apex predator and found evidence to support their hypothesis that:

> *"weakened top-down control by all elasmobranch-consuming sharks could increase abundances of their elasmobranch prey (rays, skates, and small sharks) and that the enhanced predation by these mesopredators might cascade to lower trophic levels."*

These workers pointed out that the consequences of the region-wide proliferation of rays, skates, and small sharks had cascaded down the food web through cownose rays (*Rhinoptera bonasus*) and had adversely impacted on populations of the commercially exploited bay scallop (*Argopecten irradians*) and possibly other bivalves. Myers et al. (2007) also suggested that the negative effects of the cascade may have extended to the sea-grass habitat, could have increased the stress on what was a highly

[3] Microbialization it taken to mean an increase in the microbial component of marine ecosystems.

degraded coastal benthic system and that the elimination of great sharks carried risks of broader ecosystem degradation.

Studies of the changes that have taken place in the Baltic Sea provide an example of the complex interplay between ecosystem structure and functioning, human pressure and climate. Thurow (1997) suggested that eutrophication might have led to increased fish biomass and changes in Baltic Sea fish populations have also been explained, in part, by changes in the volumes of water (and their overlaps) available to each species, squashed between hypoxic deep water and low salinity surface water. The extents of these water bodies vary with climate (changes in the flushing of the Baltic and its freshwater inflows) and eutrophication causing an increase in the dissolved oxygen demand in deep water (Vallin et al., 1999; Ojaveer & Lehtonen 2001; Koster et al., 2005). According to Casini et al. (2007, 2008) and Österblom et al. (2007), trophic cascades and ecological regime shifts have been induced by human influence on the Baltic Sea ecosystem. Using an ecosystem model, Österblom et al. (2007) showed that reduced top-down control (seal predation) and increased bottom-up forcing (eutrophication) can: *"largely explain the historical dynamics of the main fish stocks (cod, herring, and sprat) in the Baltic Sea between 1900 and 1980"*. Österblom et al. (2007) identified two major ecological transitions: (i) a shift from seal to cod dominance which was caused by the virtual elimination of marine mammals, followed by a shift from an oligotrophic to eutrophic state; (ii) a shift from cod to clupeid dominance in the late 1980s. Österblom et al. (2007) and Casini et al. (2007, 2008) concluded that shifts in the Baltic Sea were a consequence of human impact although some variation in climate may have influenced the timing, magnitude and persistence of the changes.

The third example is from the north Atlantic seaboard of the US and Canada. As a result of overfishing, the cod fishery collapsed in 1992. In 1994, the US, New England Fishery Management Council initiated plans to reduce fishing effort on ground-fish by 50% and indefinitely closed more than 4800 km^2 of Georges Bank to fisheries, which had supported one of the most diverse and prolific marine ecosystems in the world (Collins, 1994).

Removal of cod as the top predator and the resulting trophic cascade (Frank et al., 2005) combined with climate change had caused a regime shift in which cod were no longer abundant, small cod competed with forage fish and there was a substantial increase in benthic productivity, particularly shrimps and snow crab. Bundy and Fanning (2005) reported that there had been no recovery of the Scotian Shelf cod population and that the species was: "*trapped in a vicious circle: their abundance is being kept low by predation, causing an abundance so low that cod cannot compete for prey with their exceptionally abundant competitors.*"

2.2.3 By-catch and discards

Commercial fishing results in a by-catch: the capture of species that are not specifically targeted by the fishery. By-catch can include juvenile fish of the target species (which may or may not form part of the catch), non-target species of fish and other marine organisms. In the late 1990s, the global marine fisheries resulted in an annual by-catch and discard of approximately one third of landings (Mace, 1997). However, not all by-catch should be viewed in a negative way. In some countries like Norway, species caught within a fishery as by-catch are landed.

By-catch and discards have been shown to influence the population characteristics of marine mammals, sea birds, and sea turtles. According to a report published by the Marine Centre of Conservation, fishing practices including the use of bottom snagline, were among several factors that contributed to mortalities of baiji or Chinese river dolphin (*Lipotes vexillifer*) which is now considered to be functionally extinct (Turvey et al., 2007). Another example is the by-catch of Vaquita also known as the Gulf of California porpoise (*Phocoena sinus*) that is considered the most endangered marine cetacean in the world. In some cases, by-catch does not appear to have an impact on sea mammal populations. For example, Bjorge et al. (1991) reported that mortalities of 2.5% of female Hooker sea lion (*Phocarctos hookeri)* in the trawl squid fisheries off New Zealand was not thought to contribute to population changes.

The issue of mortalities of cetacean in passive fishing gears were addressed at a 1991 International Whaling Commission (IWC) Workshop where scientists noted seven of 54 species population regions (SPR) had mortality rates caused by passive gears that were considered "*to be not sustainable*" (IWC, 1991). Populations noted to be suffering from excessive mortalities included the Baiji, the Vaquita, the Indo-Pacific hump-backed dolphin (*Sousa chinensis*), two populations of bottlenose dolphins (genus *Tursiops*) off the coast of South Africa, harbour porpoise *(Phocoena phocoena*) in the Northwest Atlantic, and striped dolphin (*Stenella coeruleoalba*) in the Mediterranean.

It is evident that in the early years of the modern fishing industry, by-catch mortalities significantly reduced population sizes of "*the most commonly taken dolphins*" (Bjorge et al., 1991). However, Bjorge et al. (1991) stated that by the early 1970s, kill rates had declined and population sizes of impacted species have since stabilized. Reduction in mortalities was reported to result from: (i) training of fishing boat skippers to perform back-down procedures (i.e. not to fish when dolphins were close to the fishing boat); (ii) helping quiescent dolphins from the net; (iii) making gear modifications.

The examples summarised above are largely from non-European waters. However, sea mammals are also a by-catch in European waters. Data on the status and by-catch of cetaceans in European waters has been compiled by ICES (ICES, 2008b) and data reported by the UK from the Celtic Sea are summarised in Table 1.

Bergmann (2002) reported the results of a study of the composition of the by-catch associated with the *Nephrops* fishery in the Clyde. On average, 9 kg of discard was produced per kilogram of *Nephrops*, although the amount and composition of the discard are dependent on differences in fishing practice (e.g. type of gear used) and the area trawled.

Table 1. A summary of the 2005 and 2006 by-catch of harbour and common porpoises in the Celtic Sea. (Modified from ICES, 2008b).

Species	Gear type	Year	Estimated by-catch

			(numbers of animals)
Harbour porpoise	Gill- and tangle nets	2005	464
	Gill- and tangle nets	2006	730
Common dolphin	Gill- and tangle nets	2005	253
	Gill- and tangle nets	2006	554
	Pair trawl bass	2005	155
	Pair trawl bass	2006	40

It has been suggested that not all of the biological or ecological impacts of discards are negative. Hill and Wassenberg (1990) argued that:

> *"discarding from trawlers has the effect of transferring large quantities of biological material from the bottom to the surface. This makes food available to surface scavengers that would otherwise be inaccessible."*

In all likelihood, discarding provides ready forage for surface, mid-water, and benthic scavengers. Birds, sharks, dolphins, and other marine mammals commonly scavenge for food discarded in fishing operations. Population increases of some sea birds inhabiting the North Sea have followed changes in prey availability brought about by altered fishing patterns, especially the at-sea discarding of non-target species and processing waste. The number of scavenging sea bird pairs in the North Sea region increased from 37,000 in 1900 to 614,000 in the late 1970s (Furness, 1993; Hudson & Furness, 1988). Birds, seals, and fish were found to consume discards in fisheries located in the Wadden Sea (Berghahn, 1990). Three species of sea birds are dependent on shrimp fishery discards in Moreton Bay, Australia (Blaber & Wassenberg, 1989). Studies conducted in the Australian northern prawn fishery have shown that benthic scavengers consumed a considerable portion of the discards that reached the sea bed (see Alverson et al., 1994 citing the International Conference on Shrimp By-catch, 1992).

The increase in the populations of some species due to discards may be offset by negative impacts. For example, species that scavenge in and around fishing gear are more susceptible to incidental capture. Population enhancements resulting from discards should not be neglected when considering how to deal with discards in the future. In the case of the changes in sea bird populations in the North Sea, a strict ban on discarding may lead to substantial population changes away from conditions that have been almost a century in the making. The question that should be asked at this point however, is whether it is desirable to maintain these populations in this manner?

2.2.4 The impact of trawling and dredging

The passage of nets and ropes across the sea bed is known to impact species that live above the seabed and even fragile species within the seabed (Dinmore et al., 2003). The impact of trawling on seabed integrity (sediment cohesion and structure and benthic communities) is variable and a function of seabed composition, fishing intensity and natural disturbance by tides and wind. The north western part of the Irish Sea is intensely trawled with up to ten trawls per year (Ball et al., 2000). This may leave this area with reduced diversity and modified abundance and change in community structure from surface filter feeding species to mobile scavengers and organisms that live below the surface. However, trawling in soft-bottom habitats has been reported to have a minimal effect on the abundance and composition of the communities in this type of habitat (Gibbs et al., 1980; Butcher et al., 1981; Van Dolah et al., 1991; Bergman and Hup, 1992). Otter trawling removes target and non-target species and otter boards plough furrows in the seabed that can last for weeks or even years (Jones, 1992; Smith et al., 2000; Sanchez et al., 2000). More extensive damage has been reported to occur from trawling hard-bottom habitats (Van Dolah et al., 1987; Munro, 1992). Hutchings (1990) found that trawling physically removed or damaged much of the macrobenthic fauna and Kaiser and Spencer (1994) pointed out that trawling decreased the density of soft corals and hydroids. Service and Magorrian (1997) reported that areas of Strangford Lough (Northern Ireland) had been heavily impacted by trawling for queen scallop (*Aequipecten opercularis*) and Tuck et al. (1998) argued that trawling had clear long-term effects on the topography of the seabed and the composition of species. Reiss et al. (2009) showed that, even in habitats that were already heavily impacted, increased fishing pressure can cause additional habitat modification. Churchill (1989) stated that sediment resuspension by trawling in the Middle Atlantic Bight can be the primary source of suspended sediment over the outer shelf and that resuspension made a sizeable contribution to the total suspended sediment load over the heavily trawled central shelf area of Nantucket Shoals.

2.2.5 Competition for resources

Some commercial harvested fishes are also prey items for seabirds and mammals and the impact of fishing on the availability of prey has become an important matter in the conservation of some species. A considerable amount of research into the interaction between fisheries and wildlife has been conducted in the North Sea (Furness, 1990, 1999, 2002; Harris & Wanless, 1997; Rindorf et al., 2000; Jennings et al., 2001). The relationship between sandeel (*Ammodytes tobianus*) and black-legged kittiwake (*Rissa tridactyla*) is one much discussed example. Sandeels are an important prey of kittiwakes (Furness et al., 1996, 2002), especially during the breeding season (Frederiksen et al., 2005; Wright, 1996; Lewis et al., 2001b). Sandeels are also the target of an industrial fishery, with the sandeel managed as a large single North Sea stock and much smaller coastal stocks around Shetland and off the northwest coast of Scotland (ICES, 2003). Naylor et al. (2000) reported that:

> "*In the North Sea, overexploitation of many capelin [sic], sandeel and Norway pout stocks..... have been implicated in the decline of other wild fish stocks, ... and the distribution, populations size, and reproduction success of various seals and seabird colonies*".

Research has shown, however, that the number of North Sea seabirds has not decreased and that dramatic reductions in breeding success could not be attributed to overexploitation of food-fish by industrial fishing (Tasker et al., 2000; Yodzis, 2001).

Despite these findings, an index of kittiwake breeding success has been included in the management plan for the North Sea industrial sandeel fisheries (Furness & Tasker , 2000).

2.2.6 Other human pressures on marine ecosystems

Other human pressures on coastal and shelf sea ecosystems include those associated with contaminants, aggregate extraction, shipping, noise and litter. Some of these (e.g. contaminants) have been the subject of much study (see Gibson, 2011) for a review of the situation in coastal waters of Northern Ireland) and legislation. However, for some pressures such as litter the long-term impacts are less certain. In the case of litter, injury and death can be caused by entanglement of sea mammals and reptiles, and birds in discarded nets and ropes. There is increasing concern over the impact of microplastics that can be mistakenly ingested as food items by birds and mammals and microscopic sized plastic particles that are ingested by filer feeding organisms (OSPAR, 2009).

2.2.7 Interactions between human pressures

There are a number of examples that illustrate how the effects of human pressure from one activity can interact with the effects of a different human pressure or influence the human use of other ecosystem resources. Some of these interactions may not be negative. For example, it has been suggested that in the Baltic Sea, eutrophication supported increases in the populations of fish (Thurow 1997). In contrast, fishing activity in Chesapeake Bay (USA) resulted in a positive feedback which increased the undesirable consequences of anthropogenic nutrient enrichment. According to Jackson et al. (2001) despite intensive land use and human settlement, bottom water anoxia was not widespread in the Chesapeake Bay before the 1930s but became more widespread following the collapse of the oyster fishery. Jackson et al. (2001) argued that the filtering capacity of the oyster populations (perhaps equivalent to filtering the water in the bay every 3 days) helped maintain water quality and it was only when mechanical harvesting brought about a collapse in oyster populations that eutrophication became more problematic.

The trophic cascade resulting from shark fishing and region-wide proliferation of rays, skates, and small sharks noted earlier, resulted in a reduction in bay scallop populations (Myers et al., 2007) and had a negative effect on the commercial exploitation of bay scallops.

3. Managing Ecosystem Services: An Ecosystem Approach

3.1 Introduction

Since the 1990s, a number of ideas and approaches have been proposed to move away from the sector by sector management of ecosystem services to '*ecosystem management*' (see Lubchenco, 1994). The United Nations Convention on Biological Diversity (UNEP, 1998) gave eight reasons why the adoption of an ecosystem approach was important (Box 1) and the Convention defined an ecosystem approach as: "*A strategy for the integrated management of land, water, and living resources that promotes conservation and sustainable use in an equitable way*"

The term ecosystem approach (EA) has also been defined by The International Council for the Exploration of the Sea (ICES, 2005) as:

> *"A 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 the marine ecosystems, thereby achieving sustainable use of ecosystem goods and services and maintenance of ecosystem integrity."*

Box 1. Eight reasons for adopting an ecosystem approach to manage ecosystem services (UNEP, 1998).

1. The ecosystem concept helps to define the appropriate management level to meet the three objectives of the Convention.
2. Functioning ecosystems are indispensable for the survival of human beings and future generations as well as the global environment, as the Convention recognizes the intrinsic value of biological diversity.

3. Biological diversity is inextricably linked to ecosystem processes, functioning and resilience.
4. Ecosystem understanding allows effective and sustainable use.
5. People frequently move among ecosystems, and often use different ecosystems to satisfy their needs.
6. Humans are frequently seen as external to ecosystems even when they are residents within them.
7. The ecosystem approach allows the use of both indigenous and local knowledge, innovations and practices including traditional management systems and scientific thinking.
8. Appropriate emphasis on the range of goods, services and information which ecosystems provide to humanity.

A scientific consensus statement prepared by U.S. scientists and policy experts (McLeod et al., 2005) defined ecosystem based management for the oceans as:

> "*an integrated approach to management that considers entire ecosystems, including humans. The goal of ecosystem-based management is to maintain an ecosystem in a healthy, productive and resilient condition so that it can provide the services humans want and need.*"

Recently, the implementation of an ecosystem approach has been given impetus by shifts in policy and changes in legislation in a number of countries including China (White Paper of 1998 on the Development of China's Marine Programmes, from IOC, 2007), the US (Oceans Act of 2000, according to IOC, 2007) and Australian (Australia's Oceans Policy of 1998, according to IOC, 2007). The UK Marine Science Strategy (Defra, 2010) identified an '*ecosystem approach to management*' as being central to its vision for: "*Clean, healthy, safe, productive and biologically diverse oceans and seas*". The European Union Marine Strategy Framework Directive (OJEU, 2008) states that by:

> "*applying an ecosystem-based approach to the management of human activities while enabling a sustainable use of marine goods and services, priority should be given to achieving or maintaining good environmental status…….*".

The US consensus statement (McLeod et al., 2005) also stated that: "*Ecosystem-based management differs from current approaches that usually focus on single species, sectors, activity or concern; it considers the cumulative impacts of different sectors.*"

An ecosystem approach is not ecosystem management. It is not possible to manage an ecosystem but instead manage (and where necessary reduce) human pressure on the marine ecosystem. Furthermore, the approach is not solely about trying to identify the means for greater exploitation of a resource or removing all human pressure. Finally, an ecosystem approach is not about replacing existing methodologies (e.g. reference targets in fisheries management, see Zeller & Pauly, 2007) but equally is not an excuse for 'business as usual' (Worm et al., 2006). The goal of an ecosystem approach is to manage human activities to ensure that marine ecosystems remain '*healthy*' thereby safeguarding the sustainable use of their services for future generations.

The different definitions of the ecosystem approach all identify the need to: integrate the management of human activities; base management on scientific understanding of ecosystems and their dynamics; ensure that human pressure does not impact on the '*health*' or '*resilience*' or compromise the '*environmental status*' of marine ecosystems.

The term health is increasingly used in connection with the state of marine ecosystems but in many instances the meaning is unclear. The subject was the focus of a Cefas study that included three workshops attended by experts from UK marine institutes and universities. The workshops (see also Tett, 2011) agreed a definition of ecosystem health as:

> *"an emergent property, defined as that of a system that functions well: it delivers a range of services (for humans) over a long period, it is self-maintaining, and does not negatively impact other ecosystems. A healthy system is able to resist or recover from disturbance."*

The key components that collectively contributed to a healthy ecosystem are outlined in Table 2. In the context of the Marine Strategy Framework Directive, *environmental status* is defined by 11 descriptors (Figure 2). The designation of '*good environmental status*' requires that the attributes of the ecosystem represented by each descriptor are considered to be in a 'good' state. For example descriptor 1, relates to biological diversity and requires that diversity is maintained and that the distribution and abundance of species are consistent with prevailing physiographic, geographic and climatic conditions. Descriptor 3 relates to exploited fish and shellfish and requires that exploitation of such species is within safe biological limits and that the age and size distribution of particular populations is what would be expected of a healthy stock.

The remainder of this section is focussed on fisheries management but it is important to bear in mind that a key element of the ecosystem approach concept is integrated or holistic management. We review why an ecosystem approach is considered necessary to better manage fisheries and summarise an example that incorporates consideration of ecosystem impacts into management. Some of the perceived benefits and drawbacks of the ecosystem approach are discussed. Finally in this section we consider the use of ecological quality objectives and the concept of ecosystem health in managing anthropogenic pressures and assessments of ecosystem state.

Table 2. The components of ecosystem health (From Tett et al., ms[4]).

Component	Explanation
Structure or organization	If the system is thought of as being comprised of a network of pipes, then structure is both the pipes and the joints (the components and their connections). Structure is <u>not</u> *biodiversity* as measured by number of species or related indices, but includes *functional diversity.*
Vigour	This concerns the functioning of the network. If the network is seen as a set of pipes, then vigour concerns the fluxes through the pipes and includes primary production, nutrient cycling, and balance terms or ratios such as net production or the ratio of production to respiration. It might also include 'reproductive vigour' and 'successional vigour', the potential for the biotic community to recolonize a region that has suffered disturbance.
Resilience	The defining attribute of health and emerges (in the systems sense) from structure and vigour. It is an ecosystem's capacity to deal with externally imposed pressure without fundamental change in system quality, either by resisting the pressure, subsequently recovering from its effects, or adapting to it. When this capability is exceeded then there is a *regime shift*.
Hierarchy and heterogeneity/ granularity	Here we include: (i) *subsystems* that can to some extent function independently (e.g. pelagos, benthos); (ii) the existence of spatial sub-regions that can re-seed adjacent sub-regions following disturbance. Spatial *heterogeneity* and spatial *connectance* are parts of this component; we argue that a spatially heterogeneous ecosystem is likely to be more resilient.
Trajectory	This refers to change in the 'internal description' of an ecosystem: the way in which system state, including structure and vigour, changes with time, resolved on an appropriate spatial scale. Trajectory, plotted in state variable space, and related to pressures, is both diagnostic (it allows resilience to be estimated), and prognostic (of decreasing or recovering health) - given a theory or model that allows a domain of good health (high resilience) to be located in a state space diagram.

[4] Tett P, Gowen RJ, Painting S, Elliott M, Foster R, Mills DK, Bresnan E, Capuzzo E, Fernandes T, Foden J, Geider R, Gilpin L, Huxham M, McQuatters-Gollop A, Saux Picart S, Platt T, Racault M-F, Sathyendranath S, Wilkinson M. A framework for understanding and monitoring marine ecosystem health.

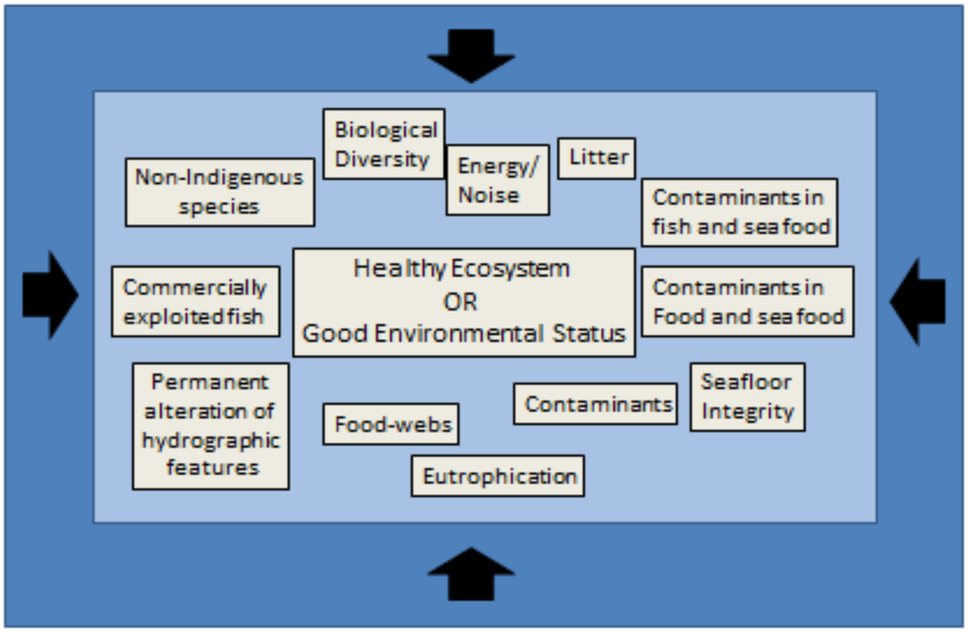

Figure 2. The eleven descriptors used to assess ecological status in the context of the Marine Strategy Framework Directive. Evaluating the influence of climate change on ecosystem state (black arrows) is an important part of the assessment process.

3.2 Fisheries science and management

The problems associated with managing marine fisheries are not new and are well-documented (Wooster, 1988; Hilborn & Walters, 1992; Ludwig et al., 1993; Smith et al., 1993; Lane & Stephenson, 1997). Examples include the collapse of sardine (*Clupeidae*) fisheries in the 1950s (Radovich, 1982 and references therein) and of Atlantic cod stocks on the eastern Scotian Shelf of Canada (Bundy & Fanning, 2005). In the North Sea, cod has been fished for hundreds of years but in the 1990s was considered close to collapse (Cook, 1998) and according to recent advice from ICES[5] is currently fished at a level that will not: "*give the highest catches in the long-term*".

[5] http://www.ices.dk/committe/acom/comwork/report/2012/Popular/cod-347_popular.pdf (accessed September 2012)

Mace (1997) suggested that a popular characterization of the status of the world's fisheries used by the media and some scientists is that *"they are on the brink of disaster"* and that the general impression gained from a review of the scientific literature is that: (i) stock collapses are common; (ii) stock collapses are almost always a result of overfishing; (iii) professional fisheries managers are frequently unable to manage the resources they control. Such impressions are not helped by the use of emotive language such as *'ecological meltdown'* (Thurstan & Roberts, 2010) in some scientific publications.

In the 1990s, 70% of marine capture fisheries were classified as fully or overexploited (Garcia & Newton, 1997) and the world's fishing fleets operated at a net loss of ~US $54 billion (Mace, 1997). Pauly et al. (2002) reported that global catches had declined since the late 1980s. In 2002, the world summit on sustainable development (Rio plus 10) recognised the problem of overfishing and issued a joint communiqué (see Russ & Zeller, 2003) calling for the restoration of depleted world fisheries by 2015. More recently, Worm et al. (2006) projected that all wild fish stocks will collapse by 2048. Pauly et al. (1998) argued that the decline in the ground-fish resource on the Northeast USA shelf combined with restrictive management resulted in a predictable scenario of serial depletion and that a reduction in the average trophic level of the northwest Atlantic catch was a response to the scarcity of higher-level predators. The European Environment Agency reported[6] that for the NE Atlantic (outside the Baltic Sea) between 30% and 54% (62% in Iberian Peninsula waters) of the assessed commercial stocks were fished beyond safe biological limits (SBL). In the Irish Sea, commercial fish stocks of cod, sole (*Solea solea*), and whiting (*Merlangius merlangus*) have declined since the 1980s and are still overexploited and unsustainably fished (Gibson, 2011).

Rice (2005) suggested that one reason why the existing methods were perceived to have failed was because of an absence of short-term incentives in the face of large and

[6] http://www.eea.europa.eu/data-and-maps/indicators/status-of-marine-fish-stocks/status-of-marine-fish-stocks-8 (accessed September 2012).

immediate social and economic impacts of displaced fishers. Rice (2005) also stated that the global crisis in food security was a further issue, referring to Watson (2002) who showed that globally, 800 million people were under-nourished. According to many, the core problem in fisheries is the imbalance in the immediate costs (short term) versus benefits (long term) because the long term economic benefits of yields from rebuilt stocks are not computed.

Despite these evident failures to manage the human pressure on these fishery resources it is equally evident that not all stocks are over exploited. Hilborn (2007) argued that some fish stocks were not headed towards extinction or even being overfished. The status of barndoor skate (*Dipturus laevis*) in the US (Boelke et al., 2005) and Canada (Kulka et al., 2002) is one example. Hilborn (2007) further argued that: "*we haven't run out of higher tropic levels yet*" and that the catch of high trophic fish only seemed to decline, because the rapid increase in lower level fish catches causes the reported mean trophic level to decline. However, Hilborn (2007) also made it clear that his statement did not mean that: "*many of the world's fisheries are not seriously overexploited and that overfishing is indeed a serious problem*". Garcia and Newton (1998) suggested that when all species were considered together, the world's fisheries resources appear to be exploited at about the maximum sustainable yield (MSY). They argued that this meant there was still room for the expansion of some individual fisheries. However, such global overviews tend to obscure local and regional issues of overexploitation.

Myers & Cadigan (1995a) were of the opinion that in most cases marine fishery resources were likely to be able to recover provided fishing pressure was reduced sufficiently. According to Mace (1997) the introduction of management measures can result in the recovery of commercially exploited fish populations and previously depleted stocks were recovering. However, as data from the eastern Scotian shelf show recovery is by no means certain even when fishing pressure is removed (Bundy & Fanning, 2005). Some locally depleted fish stocks have been rebuilt (e.g. herring (Clupeidae) stocks in Norwegian waters) since the 1990s (Mace 1997) and a recent increase in North Sea cod populations has been reported by ICES[7]. Furthermore, Hilborn (2007) suggested that although levels of fishing for jack mackerel (*Trachurus symmetricus*), anchovy (*Engraulidae*), and Pacific whiting (*Merluccius productus*) would keep stocks below the 'pristine state', the ecosystem as a whole retained its basic structure. This suggests that fisheries can be managed to minimise the impact on ecosystem structure and functioning.

The Northern Ireland State of the Seas report (Gibson, 2011) states that haddock (*Merlanogrammus aeglfinus*) spawning biomass and the biomass of herring (*Clupea harengus*) had both increased and that the Irish Sea plaice (*Pleuronectes platessa*) stock was being harvested sustainably. The *Nephrops* population in the western Irish Sea has sustained harvests since the 1970s and during this period the size composition and sex ratio have been stable, suggesting a sustainable fishery.

There are complex often inter-related reasons for the failure to manage fish stocks (Ludwig et al., 1993). Figure 3 illustrates the inter-relationships and linkages between social and economic drivers, governance and the aspiration for the integrated (holistic) assessment of the pressures that arise from human use of ecosystem services. From a scientific perspective, the models that have traditionally been used to assess fisheries and to set catch limits require data on the state of the stock and removal of fish from stocks, both of which can be imprecise. In the case of cod off Newfoundland and Labrador, and Orange Roughy (*Hoplostethus atlanticus*) off New Zealand, for example, the decline in stocks was attributed in part to overly optimistic assessments and lack of basic knowledge of population dynamics (Mace, 1997).

There is widespread awareness that increases in fishing-fleet capacity has been one of the main threats to long-term sustainability of marine capture-fishery resources and for the fisheries themselves. Russ and Zeller (2003) cited the 2002 world summit on sustainable development and suggested that the problem of overfishing was due to overcapacity and the systems of subsidies to fishing industries. According to Hilborn (2007), overexploitation was primarily the result of competing pressure for sustained

[7] http://www.ices.dk/committe/acom/comwork/report/2012/Popular/cod-347_popular.pdf (Accessed September 2012).

employment and continuation of fishing communities as well as poor governance. Murawski (2000) was of the opinion that economics would always be the primary concern for management.

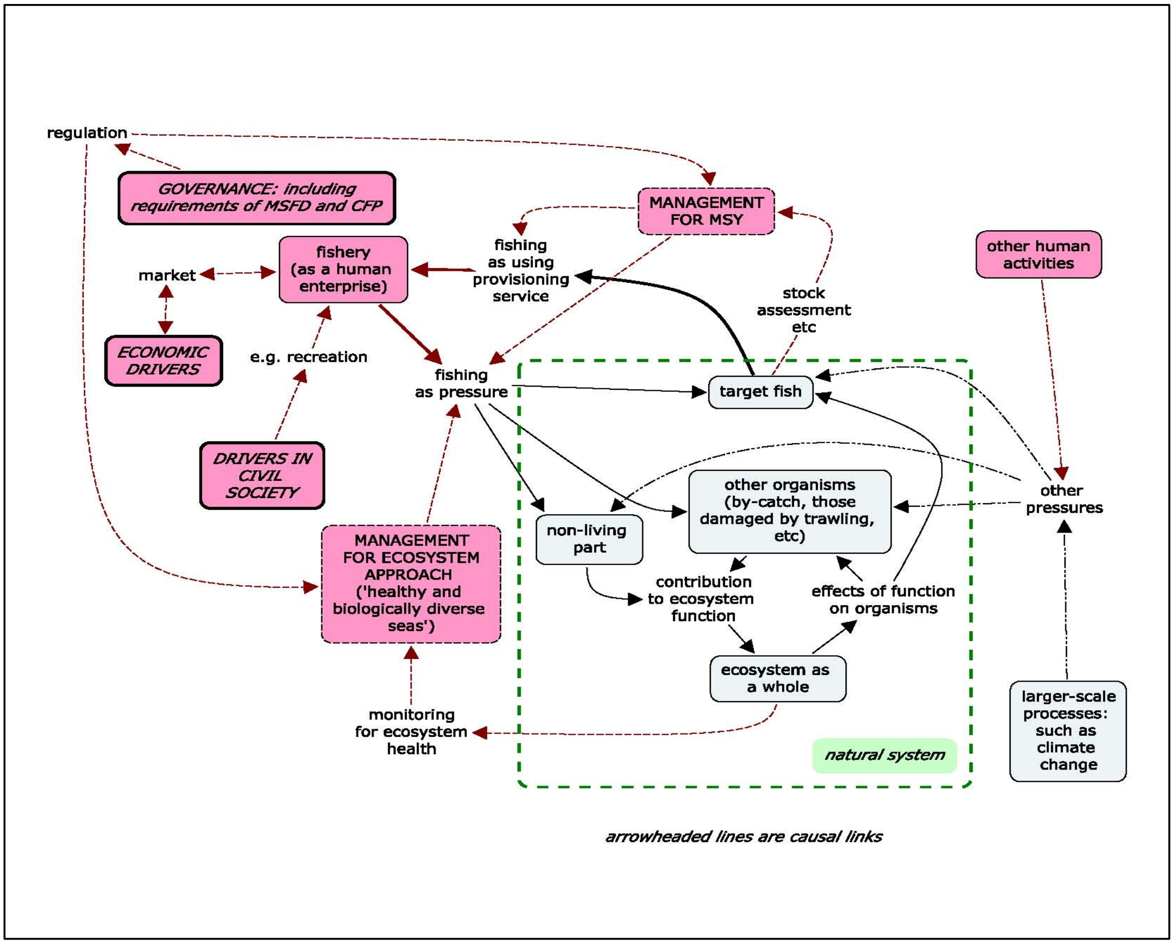

Figure 3. The interrelationships between governance, economic and societal drivers and the ecosystem approach to managing human pressures on the marine ecosystem.

Rice (2005) argued that improvements in fishing gear technology were constantly working against the objective of capacity reduction. For example, between 1992 and 1997, Canada spent approximately Can $3.9 billion in the Atlantic Fishery Adjustment package, with explicit goals of reducing the dependence of coastal Atlantic Canada on fisheries and decreasing capacity. When reviewed (Government of Canada, 1997), it was concluded that vessels had been decommissioned and while many fishermen had undertaken training the training was in new fisheries technologies. The net result (after the Atlantic Fishery Adjustment package) was a 60% increase in the functional fishing capacity of the fleet as a whole. However, technical measures can also be more selective. They provide a means of reducing by-catch of non-target species and reducing the effects of bottom trawling on the benthos.

The foundation of modern fisheries science had been established by the late 1950s and included work by Thomson and Bell (1934) on yield-per-recruit, Graham (1935) on the "law of fishing", Schaefer (1954, 1957) on stock production models, Ricker (1954) on stock and recruitment, Gordon (1954) on the economic theory of common property resources, Beverton and Holt (1957) on dynamic pool models, stock and recruitment, and other topics. Virtual population analysis (VPA) development followed in the 1960s and 1970s. In the 1990s, alternative stock assessment approaches such as size-based methods (reviewed by Gallucci et al., 1996), non-equilibrium stock production model (e.g. Prager, 1994) and stock analysis (Methot, 1990) have been used in relation to some fisheries.

Mace (1997) considered that the scientific basis for managing fish stocks was better than the popular portrayal. Myers & Cadigan (1995b) demonstrated that the global syntheses of important population processes was a useful aid to fisheries management and Froese & Pauly (1996) launched one of the most important fisheries databases, ICLARM's FishBase (supported by the European Commission since 1989) that holds a wide range of statistics from fish stocks throughout the world. Enhanced data collection programmes have enabled the application of some multi-species assessment techniques, such as multi-species VPA (Virtual Population Analysis) which has been extensively applied by ICES (Mace, 1997).

According to Pauly et al. (2002), since the early 1950s, fisheries science has attempted to equate the concept of sustainability with the idea of an optimum fishing mortality, leading to some form of maximum sustainable yield. The concept of Maximum Sustainable Yield (MSY) is embedded in fisheries management and MSY has been adopted as the primary management goal by a number of international organisations and countries (Mace, 2001). The European Commission (CEC, 2006) viewed MSY as: "*the maximum yield that may be taken year after year. It is characterized by* *a level of fishing mortality that will, on average, result in a stock size that produces the maximum sustainable yield.*"

The concept of MSY derives from population growth studies first proposed by Verhulst (1845). The S-shaped logistical growth curve (Figure 4) can be used to describe the growth of a population where the rate of reproduction is proportional to both the existing population and the amount of available resource. The final sustainable maximum population size is limited by resource availability (the carrying capacity). The logistic equation that describes the S-shaped curve can be used to calculate the size of the population that has the maximum growth rate. This is equal to half the carrying capacity population and is the maximum sustainable yield population. At and above MSY there is a surplus of individuals that can be harvested while allowing the population to grow back each year and thereby providing a sustainable harvest.

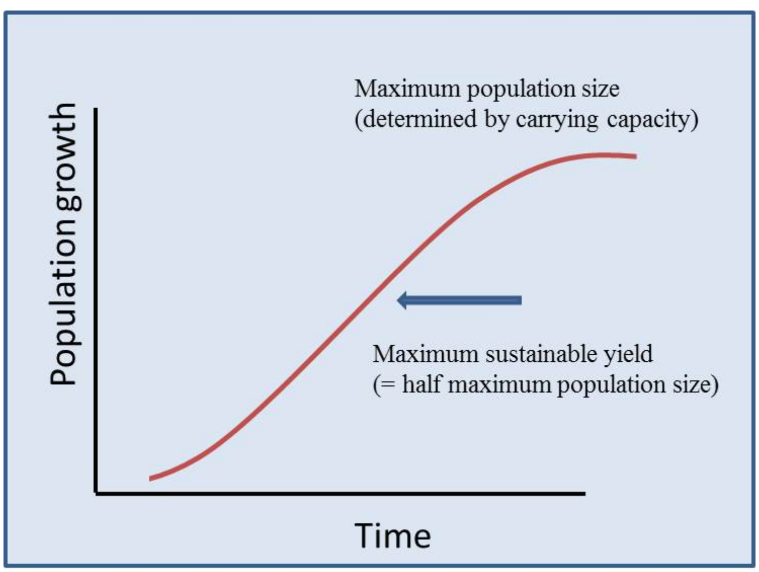

Figure 4. The logistic curve and maximum sustainable yield.

The application of MSY to fisheries management has come under considerable criticism. Holling (1973) was of the opinion that "*the goal of producing a maximum sustained yield may result in a more stable system of reduced resilience.*" That is, a system that is less able to absorb pressure. Larkin (1977) was particularly critical of MSY and stated that:

> *"In summary, from a biological point of view the concept of MSY is simply not sufficient. Nonetheless, it should be stressed that it provides a valuable rough index of production potential. As a first rough cut at management policy for major commercial species, MSY is probably acceptable. But once the level of MSY is attained, it should be expected that it may not be sustained."*

In considering the use of MSY in fisheries management, Botkin (1990) pointed out that in order to apply the MSY concept, the population in question must have a single known carrying capacity, population growth must follow the logistic curve and that the size of the population at the time of the assessment must be known. In her review of the changing attitude towards the use of MSY, Mace (2001) noted that key criticisms of MSY included poor assumptions in models and issues over data quality and the assumption of a fixed annual catch that ignores natural inter-annual variability in fish population size. Mace (2001) was of the opinion that the use of MSY has gone through three stages:

> *"(i) a fixed amount that could be taken each year indefinitely (the maximum constant yield), to (ii) the maximum average yield (or an approximation of it) that can be taken by varying catches in response to fluctuations in stock size (e.g. by fishing at a constant rate of F_{msy}), to (iii) F_{msy} being a limit to be avoided rather than a target that can routinely be exceeded."*

The latter point was also made by Witherell et al. (2000) who concluded that: *"New national legislation and international fisheries agreements are pushing fisheries management towards a new paradigm whereby MSY is treated as a limit to be avoided rather than a target to be exceeded."*

Pauly et al. (2002) also considered some of the issues around the use of MSY and suggested that although most of the models that are currently used have evolved from earlier versions that the: "*single-species assessment models and the related policies have not served us particularly well*". They, (Pauly et al., 2002) considered "*the concept of* *sustainability upon which most quantitative fisheries management is based to be flawed*" and concluded that:

> "*problems imply a need to complement our single-species assessments by elements drawn from ecology, that is, to move towards ecosystem-based management. What this will consist of is not clearly established, although it is likely that, while retaining single-species models at its core, it will have to explicitly include trophic interaction between species, habitat impacts of various gears, and a theory for dealing with the optimum placement and size of marine reserves*".

Mangel et al. (2002) also considered the utility of MSY in fisheries management and noted that: "*MSY as a management tool simply won't go away, regardless of evidence that 'managing for MSY' has not been effective.*" However, based on a modelling study, Mangel et al. (2002) concluded that "*different methods of averaging and explicitly including risk to the stock in MSY may allow MSY to be used in the context of precautionary fishery management.*"

As part of the Common Fisheries Policy (CFP) the European Commission's plan is to maintain or restore stocks to levels that can produce the maximum sustainable yield by 2015 if possible and the integration of fisheries management into an ecosystem approach to marine management (CEC, 2008). At the present time, the fisheries for which ICES provides advice have not typically been managed with MSY as an objective but future advice (ICES, 2012) will attempt to integrate a precautionary approach with the concept of maximum sustainable yield, and the ecosystem approach (see below).

3.3 The ecosystem approach concept

Lubchenco (1994) proposed a move away from sector by sector management to ecosystem management (Table 3). Sherman and Duda (1999) argued that the ecosystem approach represented a paradigm shift away from a highly focused short-term sector by sector assessment and management approach although others (e.g. Zeller & Pauly, 2007) have argued that an ecosystem approach is about adapting existing procedures. Murawski (2000) was of the opinion that an ecosystem approach to sustainable fisheries management had to include consideration of ecosystem structure and functioning (unambiguous, quantifiable, and predictive measures of ecosystem state and flux, diversity at different levels of organization); patterns of resource variability; social and economic benefits.

Hilborn (2007) argued that lessons needed to be learnt from successful fisheries and that stopping the competitive race to fish by incentives for fishing fleets together with good governance was the way forward. The aim was to apply the approach of successful fisheries to stop decline and to rebuild threatened fisheries. According to Hilborn (2007), the future of sustainable fisheries was linked to: (i) less fishing effort; (ii) lower exploitation rates; (iii) larger fish stocks; (iv) reduction in by-catch (v) increased concern about ecosystem impacts of exploitation (vi) elimination of destructive fishing practices (vii) more spatial management of fisheries (including a significant portion of marine ecosystems protected from exploitation).

Table 3. Changing the management focus (modified from Lubchenco, 1994).

From	To
Ecological	
individual species	ecosystem
small spatial scale	multiple scales
Social-ecological	
short-term perspective	long-term perspective
humans: independent	humans: integrated
Society: governance	
management split from research	adaptive management
Society: economy	
Management of commodities	sustaining production potential for goods and services

3.4 The principles and strategies underpinning an ecosystem approach

The United Nations Convention on Biological Diversity (UNEP, 1998) identified 12 principles that should underpin an EA (Box 2). Of these, number 5 and 6 specifically relate to maintaining ecosystem structure and function and managing resources within the limits of the ecosystem to provide them.

The European Marine Strategy Framework Directive (MSFD) require the EA to take account of linkages between the terrestrial and marine environment and recognise that actions on land may affect the marine environment (OJEU, 2008). The EA is based on a long-term perspective and highlights the dependence of economic and social sustainability on ecosystem sustainability and the latter is achieved by setting and achieving ecological objectives that protect ecosystem structure and function from serious or irreversible harm. For consistency it will be necessary to identify individual management regions for which ecological and operational objectives can be defined.

The European Commission (CEC, 2008), recently defined the ecosystem approach to fisheries (EAF) as a strategy that:

> *"strives to balance diverse social objectives, by taking into account knowledge and uncertainty about biotic, abiotic and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries'*

Box 2. The 12 principles underpinning an ecosystem approach to managing ecosystem services (Based on UNEP, 1998).

1. Management objectives are a matter of societal choice.
Rationale: Different sectors of society view ecosystems in terms of their own economic, cultural and social needs. All ecosystems are managed for the benefit of humans.

2. Decentralisation of management should be to the lowest appropriate level.
Rationale: Decentralized systems promote greater efficiency, effectiveness and equity. The closer the management is to the ecosystem, the greater is the responsibility, accountability and participation.

3. Understanding the effects (actual or potential) on adjacent ecosystems.
Rationale: Management interventions in ecosystems can influence other ecosystems. This needs careful consideration and analysis and decision-making;

4. Understanding ecosystems in an economic context.
Rationale: (1) The main threat to diversity lies in its replacement by alternate systems of land use. This arises through market distortions that undervalue natural systems and populations and provide perverse incentives and subsidies to favour the conversion of land to less diverse systems. (2) Alignment of incentives allows those who control the resource to benefit and ensures that those who generate environmental costs will pay.

5. Conservation of ecosystem structure and functioning.
Rationale: Ecosystem functioning and resilience depends on dynamic relationships among species; between species and their abiotic environment; physical and chemical interactions within the environment. Conservation of these interactions and processes is of greater significance for long-term maintenance of biological diversity than simple protection of species.

6. Managing ecosystems within the limits to their functioning.
Rationale: Attention must be paid to those environmental conditions that limit natural productivity, ecosystem structure and functioning and that the limits to ecosystem functioning may be affected to different degrees by temporary, unpredictable or artificially maintained conditions;

7. Use of appropriate scales.
Rationale: The approach should be bounded by spatial and temporal scales that are appropriate to the objectives. Boundaries for management will be defined by users, managers, and scientists. The ecosystem approach is based upon the hierarchical nature of biological diversity characterized by the interaction and integration of genes, species and ecosystems.

8. Long term management objectives.
Rationale: Ecosystem processes are characterized by varying temporal scales and lag effects. This conflicts with the tendency of humans to favour short term gains and benefits over future ones.

9. Management must recognize that change is inevitable.
Rationale: Ecosystems are dynamic and change, often abruptly. Adaptive management procedures must be used to anticipate and cater for such changes and should be cautious in making decisions that foreclose options.

10. Seek an appropriate balance between conservation and use of biological diversity.
Rationale: There has been a past tendency to manage components of biological diversity either as protected or non-protected. Conservation and use should be seen in context and that a full range of measures be applied in a continuum from strictly protected to human-made ecosystems.

11. All forms of relevant information (scientific, indigenous and local are useful.
Rationale: All sources of information are useful to effective ecosystem management strategies.

12. All relevant sectors of society and scientific disciplines should be included.
Rationale: Most problems of biological diversity management are complex and should where appropriate involve expertise and stakeholders at local, national, regional and international level.

Fisheries management advice from ICES (ICES, 2012) attempts to integrate a precautionary approach with the concept of maximum sustainable yield, and the EA. Sherman and Duda (1999) argued that the EA required a change in emphasis from the management of commodities to the sustainable use of the production potential of ecosystem services and proposed a modular approach based on:

- Productivity
- Fish and Fisheries
- Pollution and Ecosystem health
- Socio-economic conditions
- Governance

However, in adopting such a modular approach there is a danger that each module is treated separately. This would militate against the principle of integrating all human pressures that underpins the ecosystem approach.

Guidance on applying the ecosystem approach was prepared by ICES in 2005. The report (ICES, 2005) set out seven steps (Figure 5). The first 5 steps are largely scientific although knowledge of relevant policies and human activities is required. Step 1 involves scoping the current situation, assessing the current ecological status, evaluating relevant policies, compiling an inventory of human activities and evaluating relevant economic and social policies. Steps 2 and 3 are about comparing the current situation with the '*vision*' (e.g. *Clean, healthy, safe, productive and biologically diverse oceans and seas*: UK Marine Science Strategy, (Defra 2010); *diverse and dynamic oceans and seas which are clean, healthy and productive*: MSFD, Directive 2008/56/EC) and identifying all potential pressures associated with the human use of ecosystem services. Ecological objectives are established at Step 4. Step 5 requires the establishment of operational objectives and steps 6 and 7 concern on-going management and review.

The 11 descriptors of the Marine Strategy Framework Directive (Figure 2, p. 20) also provide a framework for the establishment of ecological objectives to achieve good environmental status.

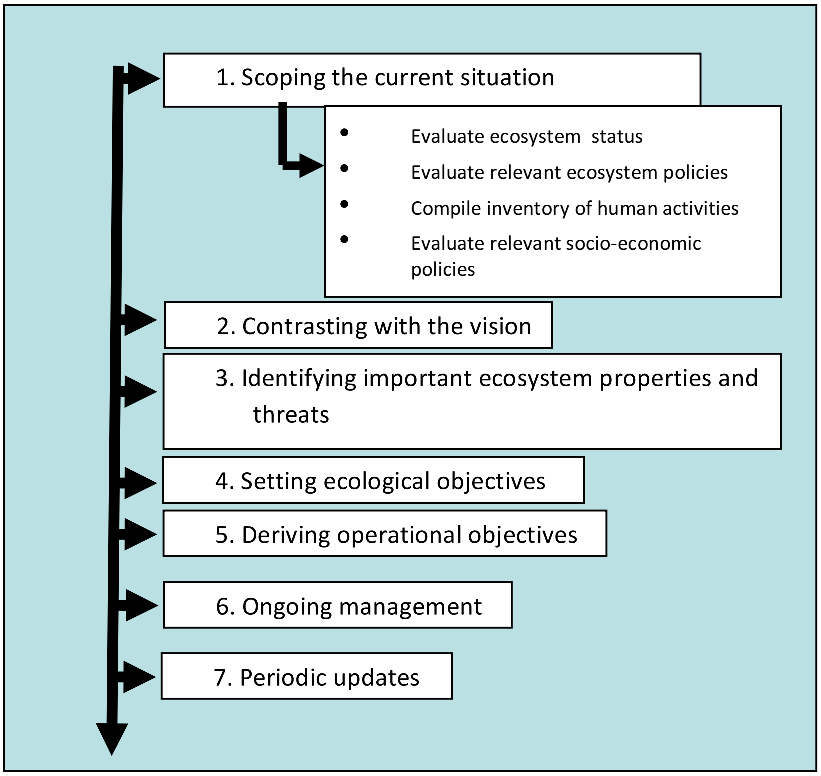

Figure 5. The seven steps involved in applying an ecosystem approach to managing ecosystem services (from ICES, 2005).

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4. Applying the Ecosystem Approach to Fisheries Management

4.1 Introduction

As argued by Zeller & Pauly (2007) an ecosystem approach is not about replacing existing methods (e.g. harvest targets in fisheries management) but clearly it should not be an excuse for 'business as usual' (Worm et al., 2006). The global shift in policy towards the ecosystem approach has been viewed as the inevitable response to the growing human pressure on marine ecosystem services (Kenny et al., 2009). The Reykjavik Declaration, Bergen Declaration, and Koge Conference are two examples of where international commitments to the EA have been made (Rice, 2005). However, Sainsbury et al. (2000) suggested that in such agreements the objectives for building consideration of ecosystems into fisheries management are often broadly stated and difficult to incorporate directly in management plans. Nevertheless, some regional, national and transnational governments are now beginning to implement the ecosystem approach.

4.2 The North Pacific ground-fish fishery

One example of the application of the EA to fisheries management was developed by the North Pacific Fishery Management Council for the North Pacific ground-fish fisheries. The principals are outlined in Table 4. According to Witherell et al. (2000).

Table 4: The North Pacific Fishery Management Council's draft ecosystem-based management policy (from Witherell et al., 2000).

Definition Ecosystem-based management is a strategy to regulate human activity towards maintaining long-term system sustainability (within the range of natural variability as we understand it) of the North Pacific.
Objective To provide future generations the opportunities and resources we enjoy today.
Goals 1. Maintain biodiversity consistent with natural evolutionary and ecological processes, including dynamic change and variability. 2. Maintain and restore habitats essential for fish and their prey. 3. Maintain system sustainability and sustainable yields of resources for human consumption and non-extractive uses. 4. Maintain the concept that humans are components of the ecosystem.
Guidelines 1. Integrate ecosystem-based management through interactive partnerships and other agencies, stakeholders, and public. 2. Utilize sound ecological models as an aid in understanding the structure, function, and dynamics of the ecosystem. 3. Utilize research and monitoring to test ecosystem approaches. 4. Use precaution when faced with uncertainties to minimize risk; management decisions should err on the side of resource conservation.
Understanding 1. Uncontrolled human population growth and consequent demand for resources are inconsistent with resource sustainability. 2. Ecosystem-based management requires time scales that transcend human lifetimes. 3. Ecosystems are open, interconnected, complex, and dynamic.

The main ecosystem consideration was the adoption of a precautionary approach to minimise impacts (on seafloor habitats and other ecosystem components – marine mammals and birds). Three targets are set for each stock:

(1) Over fishing level (OFL)
(2) Acceptable biological catch (ABC)
(3) Total allowable catch (TAC)

The OFL is regarded as unacceptable (a threshold not to be exceeded) and the ABC and TAC are, with respect to target fish and other species of human interest, regarded as acceptable from a biological perspective. By-catch controls are designed to reduce the impact on other commercial fish rather than the ecosystem although all fish caught as by-catch is counted towards the TAC. The policy includes the use of conservation zones and specifically prohibits or requires reduced fishing effort in certain areas as a conservation measure to protect species such as Steller sea lion (*Eumetopias jubatus*) and Pacific Walrus (*Odobenus rosmarus*). Further protection is afforded marine mammals by limiting the catch of their prey fish. For example, in 1997, regulation was introduced that prohibited directed fishing for forage fish species such as Capelin (*Mallotus villosus*) and other forage species, including krill (euphausiids), which are important prey for higher trophic levels.
Finally, regulations have been introduced that were designed to reduce the mortality of mammals and seabirds. Non-target limits have been set for Steller sea lions and short-tailed albatross (*Diomedea albatrus*) (Hall & Mainprize 2004).

Witherell et al. (2000) were of the opinion that the measures implemented by the North Pacific Fishery Management Council appear to be achieving the desired objectives but cautioned that ecosystem based management was an adaptive process that required periodic evaluation and modification to incorporate new scientific information. (Witherell et al., 2000) concluded that:

> *"Ecosystems are not static, and human impacts also change with technology and continued population growth. Ocean conditions can cause significant, rapid, and sometimes, unexpected changes in ecosystem components. Because still little*

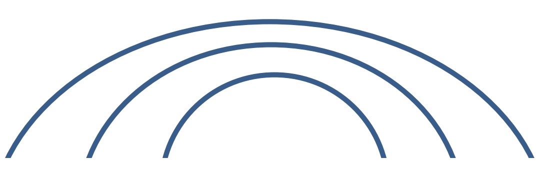
is known about marine ecosystems, an adaptive and precautionary approach is essential for all fishery management policies."

4.3 Fisheries management advice from ICES

The advice from ICES (ICES, 2012) attempts to integrate a precautionary approach with the concept of maximum sustainable yield, and an ecosystem approach. The advice states that: "*Populations need to be maintained within safe biological limits according to a precautionary approach to make MSY possible. However, within safe biological limits, an MSY approach is necessary to achieve MSY*." The advice also states that: "*In a sense, a precautionary approach is a risk-averse concept intended to avoid unproductive situations while an MSY approach is intended to make the best use of the ecosystem productivity. A precautionary approach (PA) is a necessary, but not a sufficient condition for MSY."*

The requirement for a precautionary approach is because of a recognition of the limitations of the models used to estimate MSY and that marine ecosystems are dynamic: "*fish stocks will not only change in response to the fisheries directly targeting them but also to changes in fishing patterns and fishing pressures on their prey or their predators*." Fish populations might also be expected to respond to long-term changes in marine ecosystems and natural (Ried et al., 2001) and human pressure/climate induced (Bundy and Fanning, 2005) regime change.

Limitations imposed on fisheries because of considerations of ecosystem structure and functioning have typically not been used in management policies in the ICES area (ICES, 2012). Nevertheless, ICES (2012) has stated that as the MSFD is implemented, restrictions may be required to achieve GES, especially in relation to MSFD descriptors for biodiversity, sea floor integrity, and foodwebs. ICES viewed the integration of a precautionary approach with maximum sustainable yield and an ecosystem approach as nested boundaries to harvesting living marine resources, where the outer boundary is defined by a precautionary approach to maintain fish stock productivity (Figure 6).



Ecosystem Impact
Boundary

MSY

PA exploitation boundary

Figure 6. Limits to the exploitation of fisheries based on an integration of the precautionary approach, MSY and the ecosystem approach. (Redrawn from ICES, 2012).

4.4 Non-target species reference points and marine protected areas

The use of reference points for non-target species and marine protected areas (MPAs) are both considered to be important tools in reducing the ecological effects of fishing. Historically, most regulatory effort has concentrated on commercially important species groups and it is only recently that non- target species have been considered as part of an ecosystem approach to fisheries management. Reference points can take the form of rearing success (chicks fledged per pair) as in the case of kittiwakes in the North Sea or as a limit on by-catch. For example, The Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR, 2010) has established an ecological quality objective (EcoQO) for North Sea harbour porpoise by-catch as: "*Annual by-catch levels should be reduced to levels below 1.7% of the best population estimate*." Hall (1999) considered the question of whether using reference points for a range of non-target species would achieve conservation of ecosystem integrity but the answer is no because as Hall (1999) admitted, such an approach would not encompass the range of impacts that fishing has on marine ecosystems.

Walter et al. (1999) were of the opinion that the growing interest in the use of MPAs as a tool for protecting by-catch species, allowing populations of target species to rebuild and damaged habitats to recover, was because of a realization that the predictive capability of single species fisheries assessment methods (and the management approaches they imply) will always fail to protect by-catch. Marine protected areas (MPAs) with no-take reserves combined with strongly limited effort in remaining fishable areas have shown to have positive effects in helping to rebuild depleted stocks (Roberts et al., 2001) and Pauly et al. (2002) argued that zoning oceans into un-fished marine reserves and areas with limited levels of fishing effort would allow sustainable fisheries that were based on resources embedded in functional, diverse ecosystems.

5. Outstanding Issues and Challenges

5.1 Introduction

Much of the debate about the failure to manage marine resources has focused on governance and socio-economic matters. Some of these are highlighted in Box 3, but their detailed discussion is beyond the scope of this report. However, we recognise that successful management requires careful consideration and integration of these matters with those related to the health of marine ecosystems.

Box 3. Requirements for successful management of fisheries.

The Australian Fisheries Management Act 1991 guidance (WWF-Australia, 2012).

- Efficient and cost-effective fisheries management;
- Exploitation of fisheries resources in an ecologically sustainable manner and in particular having regard to the impact of fishing on non-target species and on the marine environment;
- Maximising economic efficiency in the exploitation of fisheries resources;
- Achieving government targets in relation to the recovery of costs from the fishing industry;
- Fisheries managers are the focal point and driving force of successful fisheries management.

Elements for successful fisheries management (From Bowen, 1997).

- Provision of appropriate legislation which sets policy directions and establishes broad objectives;
- Recognition of fishing industry as part of corporate management, and definition of the role of industry in providing management advice;
- Establishing appropriate arrangements for integrating process, such as the appointment of management advisory committees, to bring together the relevant expertise for discussions and preparation of advice to decision making authority;
- Establishing appropriate administrative structure for fisheries management, and definition of the role of industry in providing management advice.

The challenges facing fisheries management (From Mace, 1997).

- Reduce fleet capacity and dependence on wild fish stocks;
- Define and implement workable access rights systems;
- Implement the precautionary approach;
- Expand aquaculture without destroying natural environments;
- Develop effective policies and institutions to achieve the preceding objectives;
- Commitment by all participants to continuous improvement in fisheries management process including regular performance reviews;
- Ongoing examination, by all participants involved in fisheries; management of their roles with a view to continuous improvement in their contribution to the corporate decision-making process;
- Ongoing examination of the agenda required for sound fisheries management, including the concepts of ecologically sustainable development and environmental policy.

It is evident from the scientific literature that much of the debate about the ecosystem approach has been focussed on fisheries. For example, Pikitch et al. (2004) suggested that: "*Ecosystem-based fishery management (EBFM) is a new direction for fishery management, essentially reversing the order of management priorities to start with the ecosystem rather than the target species*." (see also Blanchard et al., 2010). Marasco et al. (2007) were of the opinion that: "*Ecosystem-based fisheries management recognizes the physical, biological, economic, and social interactions among the affected components of the ecosystem and attempts to manage fisheries to achieve a stipulated spectrum of societal goals, some of which may be in competition.*"

There appears to have been much less discussion on using the concept of the EA to manage the collective pressures arising from the human use of marine ecosystem services. Furthermore, the elements that are required for EBFM appear much the same as for other pressures. Marasco et al. (2007) identified seven elements (Table 4) that they considered to be specific to EBFM (and distinct from the elements considered in single species management) but in our opinion the elements listed in Table 4 are relevant to other human pressures such as anthropogenic nutrient enrichment. There is a risk that in focussing on fisheries, ecological impacts arising from fisheries and other human pressures will continue to be quantified on a sector by sector basis and this will militate against the expectation that human pressures will be assessed collectively (holistically). For example, in assessing good environmental status, to fulfil the terms of the MSFD: ".......*ensuring that the collective pressure of such activities is kept within levels compatible with the achievement of Good Environmental Status*... "Similarly, ICES (2005) refers to: "*comprehensive integrated management of human activities...*" and the European Commission (CEC, 2008) has concluded that: "*Fisheries management can contribute to an ecosystem approach, but an ecosystem approach to marine management must integrate all economic sectors which impact the marine ecosystem.*"

Table 5. The seven elements that characterise ecosystem based fisheries management. (From Marasco et al., 2007).

Element
1. Ensure the broader societal goals are taken into account
2. Employ spatial representation
3. Recognise the importance of climatic-oceanic conditions
4. Emphasise food web interactions and pursue ecosystem modelling and research
5. Incorporate improved habitat information (target and non-target species)
6. Expand monitoring
7. Acknowledge and respond to higher levels of uncertainty

5.2 Merging the old with the new

Turning the vision and discussion of the EA into practical solutions which are integrated and cross-sectoral in their approach (Browman & Stergiou, 2004) takes time, because many national and regional monitoring and assessment programmes are designed around sectoral policy needs and interests (Kenny et al., 2009). Mee (2005) suggested that adoption of the EA offered an opportunity to develop more integrated policies and that adaptive management schemes provided a means of developing understandable ecological quality objectives in steps that were within pragmatic political timeframes.

Pauly et al. (2002) were of the opinion that the assessment of single species for fisheries needed to be supported by *"elements drawn from ecology"* but were unclear about what should be done with respect to the latter. This is perhaps the key challenge: how to combine methods of determining and setting harvest levels with understanding of the impacts of human pressure on ecosystem structure and functioning to achieve the desired outcomes of sustainable use of marine resources and healthy and biologically diverse seas? Furthermore, how should ecosystem state (and change in state) be quantified to manage the ecological effects of human pressure in an integrated or holistic way? Within the European Union there is also the question of how the Common Fisheries Policy will be integrated with the requirements to meet good environmental status for the MSFD.

5.3 The use of non-target reference points and MPAs

Much work is required to further develop the use of both non-target species reference points and MPAs. Most studies of non-target species have been based on abundance trends in local populations rather than regional scale populations (Jennings & Kaiser, 1998) and methods are required to assess the vulnerability at the population level in relation to fishing mortality rate and the capacity of the population to withstand that mortality (Pope et al., 2000). In relation to MPAs, issues that need to be resolved include how to determine where (and perhaps when) and how large (or for how long a time period) an area should be protected. Furthermore, there is also a concern that fishing effort may become concentrated at the edges of MPAs.

With respect to both reference points for non-target species and MPAs, careful consideration needs to be given to the reason for their use. Both can be considered conservation measures but it should not be assumed that the state of the population of a non-target species can be used to gauge the state of the ecosystem as a whole, or that the conditions within a MPA can be used as reference conditions for ecosystem state.

For non-target species reference points there is the issue of establishing appropriate reference points. If as suggested by Jackson et al. (2001) coastal ecosystems have been substantially altered by the loss of most large fish and large marine vertebrates then setting reference points becomes problematic. Reference points based on notional historical population levels of one or more species assumes that ecosystems are essentially balanced and stable. However, opinion is shifting towards a view of ecosystems (especially estuarine and coastal ecosystems) as being dynamic and highly variable (Botkin, 1990; Cuddington, 2001; Gowen et al., 2012) and this makes setting reference points and targets much more difficult.

If reference points for non-target species and MPAs are established with the aim of 'restoring' the ecosystem it should be recognised at the outset that recovery back to a notional 'pristine' or 'background state' may not take place. In addition to the above argument regarding the dynamic nature of ecosystems, evidence suggests that natural ecosystems are resilient and therefore resistant to anthropogenic pressures. However, beyond a certain level of pressure there is a danger that the ecosystem will change (the analogy is one of 'falling over a cliff') and the ecosystem may only recover slowly and not necessarily to the original conditions (Figure 7).

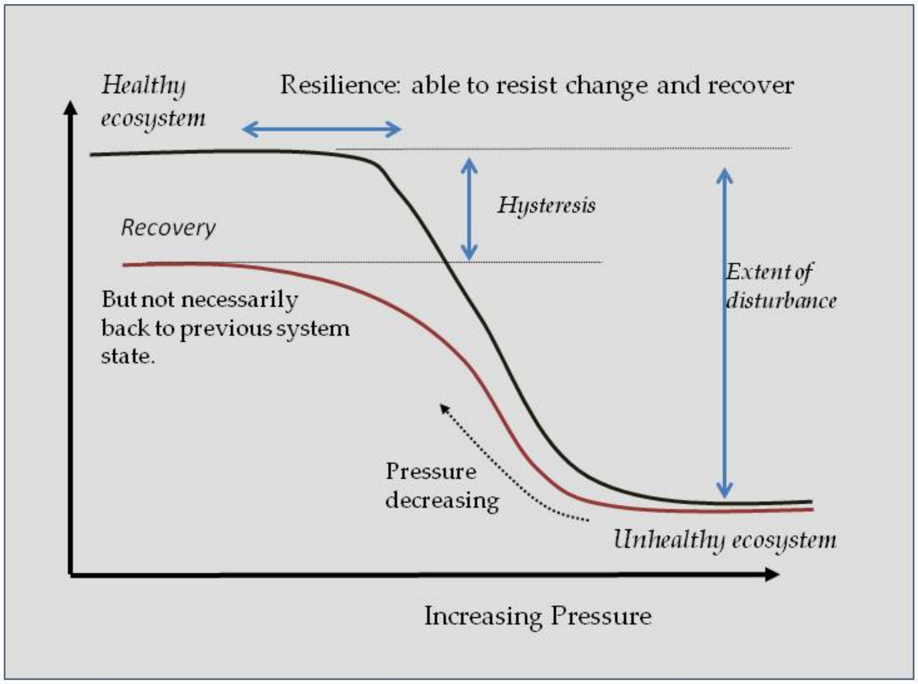


Figure 7. Changes in ecosystem state and recovery in relation to human pressure. (Redrawn from Tett et al., 2007).

5.4 Ecological Quality Objectives and ecosystem health

In a European context, the challenge is to incorporate the ecosystem effects of fisheries into the integrated or holistic management of all human pressures on marine ecosystems, while taking account of broader ecological considerations that are the subject of European directives such as those relating to habitats, biodiversity and of course, the MSFD.

Two indices have been proposed that relate specifically to fisheries pressure on marine ecosystems. According to Branch et al. (2010) the most widely adopted marine indicator is the mean trophic level (MTL) index from catches. It is intended to detect shifts from high-trophic-level predators to low-trophic-level invertebrates and plankton-feeders. The indicator underpins reported trends in human impact, declining when predators collapse (fishing down the food web) and when low-trophic-levels expand (fishing through marine food webs). However, Branch et al. (2010) pointed out that their results showed that catch MTL did not reliably predict changes in marine ecosystems, because catches were influenced by changes in economics, management, fishing technology, and targeting patterns. To detect fishing impact on the marine ecosystem greater efforts are required to measure the true abundance trends especially those most vulnerable to fishing pressure.

The Marine Trophic Index (MTI) was one of the eight indicators that the Conference of the Parties of the Convention on Biological Diversity (CBD) identified for use to: *"achieve a significant reduction in the current rate of biodiversity loss by 2010"* (CBD, 2004). The MTI is in fact the CBD expression of the mean trophic level (MTL). The latter was used together with fisheries landings by Pauly et al. (1998) to demonstrate that since the 1950s, fisheries had increasingly relied on smaller and short-lived fish and invertebrates lower down the food web (see also Pauly & Watson, 2005).

Mean maximum size (MMS) is the type of indicator that might be adopted as a measure of food web structure in reporting GES (S. Jennings pers. Com.) and has been recommended as an indicator of fishing impacts in the revision of the EU Data Collection Regulation (DCR) for fisheries. The interpretation of MMS is that high values indicate a dominance of larger species in the community.

The ecosystem approach raises important questions on how to assess ecosystem state and quantify changes in state, particularly of whole ecosystem features such as biodiversity. Powers and Monk (2009) were of the opinion that in addition to monitoring species-abundance, a suite of ecological indices was required to track ecosystem state. Similarly, Cury and Christensen (2005) stated that: *"... the objective is not to find the best indicator, but rather a relevant suite of indicators with known properties."* The way forward depends in part on our understanding of the functioning of ecosystems and the urgent need for practical methods to build greater consideration of the natural environment into the management of human use of ecosystem services.

In a review of the ecosystem concept, Tett et al (ms) suggested that there were two ways to conceptualize ecosystems. The first is empirical and based on a view that ecosystems are simply aggregates of the properties of the populations of plant and animal species and the habitats within which they live. The second is a systemic view (derived from general systems theory) on which ecosystems show a high degree of integration and interdependence and have emergent properties: features of the structure and functioning of the whole ecosystem. In other words, the ecosystem as a whole is more than the sum of its parts.

Under the empirical view ecosystem state and change in state can be assessed by monitoring a particular component or components. Even under the empirical view, however, it seems likely that the disturbance of one or more components can propagate into other parts. Considerable caution is therefore required to make sure that monitoring is not undertaken in a piecemeal fashion with a focus on those components of an ecosystem perceived to be most at risk from human activity.

Hall (1999) considered the use of emergent properties in assessing change in ecosystem state from the perspective of fisheries management but concluded that it was not possible to determine what a particular change in a "*higher order*" property might mean. Hall (1999) argued that if reference points for a range of non-target species were set "*appropriately*" then with respect to system level reference points: "*it was by no means axiomatic that we actually need them*". However, as discussed earlier, Hall's argument is based on the assumption that reference points for non-target species can be used as a proxy for the state of the ecosystem as a whole. Furthermore, the dynamic nature of ecosystems and human impacts that have already taken place may make it difficult to set reference points.

Tett et al. (ms) concluded that resilience was a key emergent ecosystem property (related to structure and function) that maintains and restores the integrity of an ecosystem in response to disturbance. Folke et al. (2004) argued that anthropogenic disturbance to ecosystems reduced resilience and increased the chances of regime shift that can, as argued earlier, bring about a change in the services that the ecosystem provides to humans. It is not possible to measure resilience directly but Tett et al. (ms) suggested that increased variability (a proxy for decreasing resilience and a general symptom of a decrease in ecosystem health) could be quantified by analysing changes in variability using a system state space approach. A further advantage of this approach is that it should be possible to relate changes in variability to all human pressures (and climate change) on the marine ecosystem thereby providing a holistic assessment.

The ideas and concepts reviewed by Tett et al. (ms) provide a valuable insight into how holistic indicators that are grounded in a theory of ecosystem structure and functioning could be developed although further work is required. However, given the urgent need to include greater consideration of marine ecosystems and their capacity to provide a range of sustainable services to human, means that we need to rely on indicators and reference points for particular ecosystem components.

The establishment of operational objectives with indicators and reference points (values associated with a particular state of the ecosystem) is part of the ICES guidance on applying the ecosystem approach (step 5 in Figure 5). Rogers et al. (2007) suggested that ecosystem objectives should be used to describe the desired or undesired state of marine ecosystems and provided a list of draft ecological objectives. One example of an objective given by Rogers et al. (2007) was for marine vertebrates (sea birds, mammals and reptiles), that is designed to: "*Maintain or restore seabird and marine mammal populations to above viable limits, taking account of natural population dynamics and trends.*" Rogers et al. (2007) and Kurtz et al. (2001) highlighted the importance of interpretation of results and utility. To be of value, indicators or Ecological Quality Standards (EcoQSs) must convey information on ecosystem state that can be used in decision making.

Tett et al. (2007) suggested that indicators of ecosystem change could be divided into five groups: bulk indicators, frequency statistics, flux measurements, structural indicators and indicator species. The first three groups relate to energy flow or vigour and coupling and the last two groups to changes in community structure. In relation to the EU Water Framework Directive, Tett et al. (2007) considered that most of these indicators allow definition of EcoQSs to provide thresholds to disturbance but that most are better seen as Ecological Quality Objectives (EcoQOs) which require indicator values to be within a defined range rather than a single value or threshold.

The Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR, 2010) has been developing a range of EcoQO systems to:

a. define in measurable objectives the 'envelope' within which the general OSPAR goal of a healthy and sustainable marine ecosystem lies;

b. give an integrated view on how the OSPAR Strategies together can deliver this general goal;

c. contribute to the development of indicators, with reference levels, targets and limits that will be required to apply the generic qualitative descriptors of GES for the MFSD at the (sub-)regional level;

d. provide tools for integrated assessments of the quality status of the OSPAR Regions for the QSR 2010 (which will contribute to the initial assessment required under the MSFD);

e. structure strategic biological monitoring using the DPSIR (Driving forces-Pressure-State-Impact-Response) model;

f. communicate objectives and increase environmental awareness.

The ecological quality objectives cover a broad range of impacts (Table 5) but most are concerned with particular components for which the link between pressure and impact is well defined. According to OSPAR (2009) the selection of EcoQOs was pragmatic: based on existing knowledge and North Sea monitoring programmes.

Table 6. Ecological quality objectives established for the North Sea by OSPAR. (From OSPAR 2009).

Ecological Quality objective
Spawning stock biomass of commercial fish species
Harbour and grey seal population trends
Harbour porpoise by-catch
Plastic particles in seabird stomachs
Contaminants in seabird eggs
Proportion of large fish in (demersal) fish community
The proportion of oiled guillemots
Imposex in dogwhelks
Eutrophication

There is, however, a question over whether these EcoQOs are sufficient to provide a holistic assessment of ecosystem structure and functioning. For example, Link et al. (2002) examined a suite of abiotic, biotic, and human metrics for the northeast US continental shelf ecosystem at the aggregate, community, and system level with more than 30 different variables measured over three decades. Their primary goals were to: (i) describe ecosystem state; (ii) improve understanding of the relationships between key ecosystem processes; (iii) evaluate potential reference points for ecosystem-based fisheries management. However, Link et al. (2002) found that focussing on too few variables was likely to result in erroneous conclusions.

OSPAR (2009) reported that development was slow (reasons included lack of data, coordination and commitment) but identified a need to further develop some EcoQOs, cross reference to the MSFD and build the integrated monitoring and assessment procedures that linked directly to management and management actions. In developing the EcoQO system further, OSPAR noted the work done by the Helsinki Commission (HELCOM) in the Baltic Sea which is an attempt at the ecological restoration of the sea. The Baltic Sea action plan included implementation of joint recommendations to curb pollution originating from land and marine sources, ensuring safer maritime traffic, and protecting biodiversity, for example, by setting up a network of Baltic Sea protected areas (Backer & Leppänen, 2008). In 2003, HELCOM adopted the EA to manage human activities and from 2005 adopted the objectives set out in the Helsinki Convention that covers: eutrophication; impacts of hazardous substances; the overall status of biodiversity (including the impact of fisheries and shipping). Backer and Leppänen (2008) further reported that the future aims of the Contracting Parties were an agreement on indicators with quantitative targets that will enable a quantitative assessment of 'good ecological status' and progress towards the goals of HELCOM, the Convention on Biological Diversity (CBD), as well European legislation concerning marine environmental protection.

The Comprehensive Procedure assessment, which includes the overall EcoQO for the North Sea and its integrated set of specific EcoQOs, currently forms a good basis to address the descriptor of good environmental status under the Marine Strategy Framework Directive. Given that the OSPAR Comprehensive Procedure which is applicable to the whole OSPAR Convention Area, the overall EcoQO and its region-specific integrated set of five EcoQO components for eutrophication may be suitable for application in the OSPAR regions beyond the Greater North Sea.

In summary, at the present time, and until there is a better theoretical understanding of emergent ecosystem properties and how such properties can be used in the ecosystem approach, there is little choice but to make use of indicators and reference points for particular ecosystem components. Nevertheless, in our opinion it is important not to lose sight of the risks of failing to identify important changes in whole ecosystem properties and the likelihood of regime shift by: (i) assuming that reference points for non-target species can be used as indicators of ecosystem health; (ii) adopting piecemeal monitoring of ecosystem state and change in state.

5.5 Assessing ecosystem state

Assessing ecosystem state is one of the first steps in the ICES advice on implementing the ecosystem approach. Hall and Mainprize (2004) suggest that recent approaches that aggregate data from a wide range of ecosystem variables to present an overview of ecosystem state may provide a framework for stakeholder dialogue on managing ecosystem services. Several techniques have been used. Ecospace, which relies on the Ecopath mass-balance approach (Walters et al., 1999), is a spatially explicit model for policy evaluation that can be used to evaluate the influence of MPAs. The use of Ecospace has shown that the potential benefits of MPAs can easily be negated by high movement rates, and especially by concentration of fishing effort at the edge of MPAs, where cascade effects generate prey gradients that attract predators out of the protected areas (Walters et al., 1999). Lees & Mackinson (2007) who developed an eco-path model for the Irish Sea highlighted the influence of model complexity on model output.

Several studies (Link et al., 2002; Kenney et al., 2009; ICES, 2010) have used multivariate statistical techniques to investigate the influence that fishing activity and climate change have on marine ecosystems. In the study by Link et al. (2002) noted above, empirical indicators of ecosystem state were examined and standard multivariate statistical methods applied to describe changes in the system. Kenny et al. (2009) found that the North Sea pelagic and benthic parts of the ecosystem were predominantly top-down (fishery) controlled between 1983 and 1993, but the pelagic component shifted to bottom-up (ecosystem) control during the following decade. Kenny et al. (2009) also found evidence in support of the need for precautionary management measures taken in relation to setting fishery quotas. Finally, the system state-space approach of Tett et al. (ms) shows promise as a method to track changes in emergent ecosystem properties such as resilience.

6. Synthesis

In this study we set out to review the concept of the ecosystem approach to managing anthropogenic pressure associated with the human use of marine ecosystem services. In this section we draw together information presented in this report by asking a series of questions.

1. What is the objective of the ecosystem approach?

Humans have been using the services provided by marine ecosystems for hundreds of years but there is a general perception that strategies to manage the human pressure associated with these services have met with little success. The lack of success has resulted in failure to achieve sustainable use of resources and has caused 'unacceptable' damage to the structure and functioning of marine ecosystem. Managing resources on a sector by sector basis (Sherman & Duda, 1999) and failure to recognise that there are limits to the capacity of marine ecosystems to provide resources (Lotze, 2004) can lead to unforeseen interactions of different pressures on ecosystems. It is also evident that changes in ecosystem state brought about by one pressure can negatively affect other ecosystem services.

The objective in adopting the ecosystem approach is the comprehensive, integrated management of human pressure on marine ecosystems thereby protecting the structure and functioning of these systems and ensuring the long-term sustainability of the services they provide. Applying the EA will require a move away from managing anthropogenic pressure on marine ecosystems on a sector by sector basis where the focus has traditionally been on single species, activity or concern over a particular component of the ecosystem.

2. Should ecosystem based fisheries management be separated from the ecosystem approach?

The impacts of fishing activity on marine ecosystems should be part of an integrated approach to the management of all human pressures on the marine environment. In our opinion the answer to this question is therefore no.

Much of the debate about the ecosystem approach has focussed on fisheries and definitions of ecosystem-based fishery management include statements such as: *"reversing the order of management priorities to start with the ecosystem rather than the target species*" (Pikitch et al., 2004) and: "*attempts to manage fisheries to achieve a stipulated spectrum of societal goals, some of which may be in competition.*" (Marasco et al., 2007). However, giving greater consideration to the ecosystem and balancing societal needs are elements of several definitions of the ecosystem approach. Furthermore, the requirements proposed by Marasco et al., (2007) to achieve EBFM are broadly applicable to the pressures associated with the use of other ecosystem services. In our opinion, such a focus on fisheries is the same as the much criticized sectoral approach and is inconsistent with the expectation of integrated management of human activities as outlined in the MSFD (OJEU, 2008) and ICES advice (ICES, 2005).

3. What are the ecological effects of marine fisheries?

The ecological consequences of commercial fishing include the effects of removing top predators, the impact of trawling on the benthos and removal of fish that are a food for marine organisms. Fisheries tend to remove large, slower-growing fish species first and these are typically the species that occupy higher trophic levels (TL). By removing top species the mean trophic level of the remaining fishes decreases and this has been referred to as '*fishing lower down the food chain*'. One consequence of this can be a reduction in the 'buffering' (i.e. resilience) capacity of marine food webs (Pauly et al., 2002). Another effect of fisheries is what is considered to be an increase in the microbial component of coastal food webs which has been termed '*microbialization*' (Jackson et al., 2001). Furthermore, the removal of top predators can influence ecosystem structure and functioning by causing trophic cascades and regime shifts.

The impact of dredging and trawling on the benthic habitat and therefore the benthic community can be severe. The use of otter trawls removes target and non-target species and the otter boards plough furrows in the seabed that can last for weeks or even years (Jones, 1992; Smith et al., 2000; Sanchez et al., 2000). Extensive damage has been reported to occur from trawling hard-bottom habitats and it has been found that trawling physically removes or damages much of the macrobenthic fauna. Some argue that trawling had clear long-term effects on the topography of the seabed and the composition of species. Others have shown that, even in habitats that were already heavily impacted, increased fishing pressure can cause additional habitat modification. Commercial fishing generally results in by-catch of non-target species and this has been reported to have influenced the population characteristics of marine mammals and turtles and sea bird. In the late 1990s the global marine fisheries resulted in an annual by-catch and discard of approximately one third of landings.

4. <u>What consideration is given to the ecosystem approach in the Common Fisheries Policy and the MSFD?</u>

Both the Common Fisheries Policy and the MSFD are underpinned by the ecosystem approach. The concept of maximum sustainable yield (MSY) viewed as the maximum that may be taken year after year (CEC, 2006) is a key element of the Common Fisheries Policy and there is an expectation that European Community fisheries will be managed according to MSY by 2015. However, the Commission's approach (FAO, 2003a), is to develop an integrated strategy for fisheries management based on the aim to:*"strive to balance diverse social objectives .... and applying an integrated approach to fisheries within ecologically meaningful boundaries."*

The Marine Strategy Framework Directive is likely to be the main pillar for European marine environmental management for the foreseeable future. The directive begins from the standpoint of marine ecosystem being a precious heritage that must be protected, preserved and where necessary restored. Furthermore, while the directive recognises the importance using natural resources such as fisheries, there is an expectation that: "*the collective pressure of such activities is kept within levels compatible with the achievement of Good Environmental Status*...". The directive has identified 11 descriptors (Figure 2) and there is an expectation that good environmental status will be achieved by 2020.

The challenge is to bring these two directives together and ensure that the way in which the future management of anthropogenic pressures on marine ecosystems delivers societal needs and maintains healthy ecosystems.

5. <u>What is the current advice from ICES?</u>

There are two strands of advice from ICES. The first relates to the ecosystem approach concept and the second is specifically concerned with advice on fisheries. Guidance on applying the EA was prepared by ICES in 2005. The report (ICES, 2005) set out seven steps (Figure 5). The first 5 steps are scientific (with required knowledge of relevant policies and human). Step 1 involves scoping the current situation, assessing the current ecological status, evaluating relevant policies, compiling an inventory of human activities and evaluating relevant economic and social policies. Steps 2 and 3 are about comparing the current situation with the vision. Ecological objectives are formulated at Step 4 and are linked to the establishment of operational objectives (step 5). Steps 6 and 7 concern on-going management and review.

Future ICES advice on fisheries will be based on an integrated approach that combines the precautionary principle, the concept of maximum sustainable yield, and the ecosystem approach (Figure 6). The requirement for a precautionary approach is because it is recognised that the models used to estimate MSY have limitations and that marine ecosystems are dynamic: fish populations are not only influences by fishing pressure changes in ecosystem structure and functioning can also influence fish populations.

At the present time, consideration of ecosystem state is not used as part of advice to set yield. ICES (2012) has stated that as the MSFD is implemented, restrictions may be required to achieve GES, especially in relation to MSFD descriptors for biodiversity, sea floor integrity, and foodwebs. Fisheries advice from ICES is set to evolve further in the near future because of the European Commission requirement to using MSY for quantifying levels of harvest. Both OSPAR and HELCOM are developing ecological quality objectives (EcoQOs) which 'read across' to the MFSD and these are a valuable contribution to the development of the ecosystem approach.

6. <u>Is there a role for non-target species reference points and marine protected areas in the ecosystem approach?</u>

The answer to this question is yes although more work is needed and careful consideration needs to be given to the reason for their use. There is growing interest in the use of reference points for non-target species and marine protected areas as tools for reducing the impact of human pressure on marine ecosystems. Reference points can take the form of rearing success (chicks fledged per pair) as in the case of North Sea kittiwakes, or as a limit on by-catch such as the OSPAR ecological quality objective for North Sea harbour porpoise which has been set at by-catch being below 1.7% of the estimate of the population.

Marine protected areas (MPAs) with no-take reserves combined with strongly limited effort in remaining fishable areas have shown to have positive effects in helping to rebuild depleted stocks (Roberts et al., 2001) and it has been suggested that zoning oceans into un-fished marine reserves and areas with limited levels of fishing effort would allow sustainable fisheries that were based on resources embedded in functional, diverse ecosystems (Pauly et al. (2002).

Reference points for non-target species and MPAs can both be considered conservation measures but it should not be assumed that the state of the population of a non-target species can be used to gauge the state of the ecosystem as a whole, or that the conditions within a MPA can be used as reference conditions for ecosystem state. For non-target species reference points there is the issue of how to establish appropriate reference points. If as suggested by Jackson et al. (2001) coastal ecosystems have been substantially altered by the loss of most large fish and large marine vertebrates, then setting reference points becomes problematic and reliance on notional historical population levels may be inappropriate. Finally, if reference points for non-target species and MPAs are established with the aim of 'restoring' the ecosystem it should be recognised at the outset that recovery back to a notional 'pristine' or 'background state' may not take place.

7. <u>How should the ecological effects of human pressures be quantified?</u>

The ecosystem approach raises important questions on how to assess ecosystem state and quantify changes in state, particularly of whole ecosystem properties such as biodiversity. Some advocate the use of a suite of ecological indices (Cury & Christensen, 2005; Powers & Monk, 2009). The establishment of operational objectives with indicators and reference points (values associated with a particular state of the ecosystem) is part of the ICES guidance on applying the ecosystem approach (ICES, 2005) and a suite of EcoQOs has been developed for the North Sea (OSPAR, 2009) although OSPAR has reported slow progress.

Most of the EcoQOs, recommend by OSPAR relate to a particular component of the ecosystem and there is a question over whether these EcoQOs are sufficient to assess whole ecosystem features and provide a holistic assessment of ecosystem structure and functioning. An alternative is to use the emergent properties of ecosystems such as resilience (related to structure and function) that maintains and restores the integrity of an ecosystem in response to disturbance (Tett et al., ms). However, further work is required to gain a better theoretical understanding of emergent ecosystem properties and how such properties can be used in the ecosystem approach.

At the present time, the best approach is to make use of indicators and reference points for particular ecosystem components. Nevertheless, since it is likely that the disturbance of one or more components can propagate into other parts of the ecosystem, considerable caution is required to make sure that monitoring is not undertaken in a piecemeal fashion with a focus solely on those components of an ecosystem perceived to be most at risk from human activity. In our opinion, it is important to recognise that such a focus may fail to identify important changes in whole ecosystem properties and the likelihood of regime shift.

8. <u>What is the next step for developing the ecosystem approach in the Irish Sea?</u>

Following the advice from ICES (2005) an appropriate next step in establishing the ecosystem approach in the Irish Sea should be a quantitative assessment of ecosystem state (Step 1 of the ICES advice). Such an assessment (based on a variety of numerical approached such as multivariate statistical techniques and the system state-space approach) would be timely given that ICES advice is evolving and that implementation of the MSFD in at an early stage.

From a fisheries perspective, a useful next step would be to build on the assessment of ecosystem state (as above) by determining the probability of change in ecosystem state and the economic consequences. Part of this would include the development of an ‘inventory’ for each stock. The elements of what such an 'inventory' might include are given below using herring as an example:

- Stock status versus MSY and trends
- Existing regulatory measures (TAC, closed areas and periods)
- Discarding/slippage
- Fishing pressures on ecosystem (including by-catch)
- Non-fishing pressures (e.g. other human pressures/climate change)
- Biological dependencies/interactions (e.g. with other fish species, e.g. sprat)
- Stock mixing with Celtic Sea herring stock (Mourne herring investigated as separately exploited sub stock)
- Habitat requirements (e.g. spawning gravels, food resource)
- Threats to habitat, e.g. gravel extraction, especially inshore gravels on the Mourne shore
- Economic value of fishery to sector and community and the societal value placed on herring
- Value of use of gravels versus the value of the fishery resource
- Extraction unsustainable/substrate irreplaceable while fishery resource sustainable

7. Recommendations

1. Assessment and management of the ecological effects of fisheries should be undertaken as part of a holistic approach to managing the human use of ecosystem services.
2. An assessment of the Irish Sea should be undertaken to: determine its current ecological state; quantify the probability of future change in state; determine the economic consequences of any future change in ecosystem state on the sustainable use of ecosystem services.
3. Further research is required to investigate how emergent ecosystem properties can be used to provide a holistic assessment of changes in ecosystem state brought about by human pressures.

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