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Cold water coral ecosystems and their biodiversity: a review of their economic and social value

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Abstract

Despite the growing scientific literature on cold-water corals (CWC) there appears to be no research of an economic nature applied to the resource. This paper presents an overview of the goods and services of CWC and their associated biodiversity. Use and non-use values associated with CWC are presented, and the methods relevant for assessing their valuation are discussed. The impact of human induced disturbance on CWC is reviewed, in order to indicate how knowledge of CWC values can be used by policy makers in the management of CWC as a habitat and vehicle for biodiversity.

Keywords: cold-water coral, goods and services, total economic value (TEV), resilience, biodiversity

1. Introduction

The conservation of cold-water coral (CWC) emerged as a significant environmental issue in the late 1990s and a number of spatial designations and management measures have been put in place globally with a view to protecting CWC ecosystems[1]. The introduction of these conservation measures indicates increasing awareness of the values found in ecosystems on the ocean floor, even though the actual protection given has been limited in extent.

It is thought that CWC ecosystems are important for a number of reasons. First, CWC ecosystems are outstanding for maintaining one of the most bio-diverse ecosystems in the deep sea (Rogers, 1999). Second, CWC are thought to provide nursery grounds and habitat for protection, reproduction and feeding for a number of species, including commercial fish species (Costello et al., 2005; Fosså et al., 2002; Husebø et al., 2002; Rogers, 1999). Third, other potential values associated with the reefs include paleo-climate indicators, source of material that may be used in the production of novel pharmaceutical compounds, as well as a sink for CO_2 sequestration (Grehan et al., 2003). Corals also hold an existence value regardless of their direct or indirect uses to society (Beaumont et al., 2007).

However, CWC habitat is faced with a number of serious threats including deep sea fishing, cable laying and oil and gas drilling (Fosså et al., 2002; Freiwald et al., 2004; Gass and Willison, 2005). Deep sea trawling is thought to represent the single biggest threat to CWC. Globally, CWC ecosystems are coming under increasing pressure from the fishing sector due to exhaustion of commercial fish stocks in readily accessible inshore waters. With increasing pressure on inshore stocks, fishermen have been forced to explore deeper waters (Anon., 2004). These new fisheries are facilitated by the development of new fishing gears and sonar technologies.

Policy makers recognise that the economic success of CWC protection measures requires information on the value of goods and services associated with CWC ecosystems. Such information would enable decision makers to focus their attention on initiatives with the greatest potential to protect CWC but at the same time safeguard marine commercial interests and livelihoods. However, because research on CWC is still in its infancy, very little information of an economic nature is available. Although studies have been carried out on law (Long and Grehan, 2002), and ecology (Freiwald et al., 2004; Roberts et al., 2006; Rogers, 1999) of CWC, as well as ecosystem goods and services in tropical reefs (Moberg and Folke, 1999) we could not find a single paper on CWC economic values. Therefore in this study we aim to:

(a) Identify the ecological goods and services associated with CWC as a habitat and vehicle for biodiversity;
(b) Review the external costs due to human induced disturbance on CWC and its related biodiversity;

1 Examples of provisions to enable CWC protection can be found in the EU Habitats Directive, the Magnuson Stevens Fisheries Conservation and Management Act (US) and the Canadian Oceans Act.

(c) Define use and non-use values associated with CWC and indicate how CWC values can be used to formulate policy for CWC ecosystems.

The remainder of the paper is organised as follows: section 2 discusses the ecology and distribution of CWC. Section 3 gives some background to the ecological goods and services associated with CWC habitat and biodiversity. Following this, section 4 presents the human induced disturbances. In section 5 the methods which can be applied to ascertain the value of CWC are outlined. Section 6 discusses the economic and policy implications of the study findings and finally section 7 concludes and makes some recommendations for further research.

2. Ecology and distribution of CWC

Corals are colonies that consist of tiny animals called polyps that are related to and look like sea anemones (Mullineaux and Mills, 2005). Cold-water corals are found in all of the world's oceans and the Mediterranean Sea (MacIsaac et al., 2001) with a global coverage that may exceed that of warm-water reefs.

Unlike shallow water corals, cold-water corals do not need sunlight to survive, because they do not rely on the symbiotic relationship with photosynthetic algae (Mullineaux and Mills, 2005) and so can live at depths below the photic zone (Hatcher and Schiebling, 2001). Deep-water reef-forming corals live in the dark and require a hard substrate on which to settle (Rogers, 1999). They prey on zooplankton and CWC are associated with strong currents which aid in the supply of food, egg and larval dispersal as well as removal of waste products. Reefs occur at depths of 39 – 3000 metres and a temperature range of 4° - 13°C (Freiwald et al., 2004).

CWC reefs grow slowly at about 4-25mm per year and they are slow to recover from damage by human activity (Freiwald et al., 2004; Long and Grehan, 2002; Rogers, 1999). Some of the larger reefs around the world are estimated to be 8000 years old. With such a slow growth rate it is thought that it will take a very long time, if ever, for damaged reefs to recover.

3. Goods and services of CWC

3.1 CWC and its genetic diversity

Although deep ocean exploration is still in its infancy, new research on genetic[2] and biological biodiversity indicates that CWC ecosystems are remarkable for maintaining some of the most species-rich ecosystems in the deep sea (Freiwald et al., 2004). There is high diversity of CWC species as well

2 Genetic diversity usually refers to the genetic variation that exists within a species (the gene pool). Genetic diversity has been proposed as the basis on which to make conservation decisions using evolutionary distinctiveness of taxa when assigning them priorities for preservation Crozier, R.H., 1992. Genetic diversity and the agony of choice. Biological Conservation, 61(1), 11-15, Solow, A., Polasky, S. and Broadus, J., 1993. On the measurement of biological diversity. Jounal of Environmental Economics and Management, 24, 60-68, Vane-Wright, R., Humphries, C.J. and Williams, P.H., 1991. What to protect? Systematics and the agony of choice. Biological Conservation, 55(3), 235-254, Weitzman, Martin L., 1998. The Noah's Ark Problem. Econometrica, 66(6), 1279-1298.

as a high diversity of species which congregate on CWC reefs. The following discussion is mainly concerned with the latter form of diversity. The most common CWC species being *Lophelia pertusa*, *Madrepora oculata* and *Oculina varicosa*. *Lophelia pertusa* reefs are considered to be as biodiverse as some shallow-water tropical coral reefs (Rogers, 1999).

CWC ecosystems are thought to offer new opportunities for pharmaceutical, engineering, medical and food research (Grehan et al., 2003; McAllister and Alfonso, 2001; Witherell and Coon, 2001). In the 18^{th} and 19^{th} centuries Norwegian fishermen collected and used deep water corals for 'powerful medicaments' (Wilson, 2001), indicating some medicinal effects from CWC. The coral species *Sarcodictyon roseum* is being used in clinical trials for the cure of various strains of cancer and bamboo corals are being used for bone grafting (Ehrlich et al., 2006).

Many species of coral are traded worldwide, the majority being of the shallow water variety (Freiwald et al., 2004; Moberg and Folke, 1999). Precious cold-water corals were harvested for jewellery in Hawaii with the use of remotely operated vehicles (ROVs) until the year 2000 when a Coral Reef Ecosystem Reserve in the North-western Hawaiian Islands was established (Grigg, 2002). Grigg (2002) documented that in the 1999 – 2000 season almost 1.5 of tonnes of CWC was harvested and auctioned for an average price of $US 239 per kg. A deepwater gorgonian coral in Alaska, *Primnoa*, is considered valuable for the jewellery market because of the 'lustre of the skeleton when polished as jewellery' (Heifitz, 2002). Future use of compounds from CWC as described above may be discovered by harvesting CWC, but scientists have also used the natural coral structure to successfully synthesise bone analogs thereby reducing further destruction of corals for this purpose.

3.2 The value of CWC and its functional diversity

3.2.1. Biotic Services and resilience

Biotic services refer to the functional[3] values associated with CWC reef biodiversity and the role of CWC as an Essential Fish Habitat in supporting specific fisheries. For example, coral grounds appear to act as a habitat for many species; including fish of commercial value (Costello et al., 2005; Fosså et al., 2002; Husebø et al., 2002; Rogers, 1999). The branches of corals act as a refuge for many deep-water species (Puglise et al., 2005). Invertebrates such as brittle stars, sea stars and feathery crinoids live directly on the coral colonies, and smaller animals burrow into the skeletons (Mullineaux and Mills, 2005).

Deep water coral reefs coincide with areas where higher concentrations of fish can be targeted. Fishermen have observed that more fish are located in coral areas than adjacent areas (Puglise et al., 2005). Redfish (*Sebastes*) are found in high abundance in *Lophelia* reef areas (Fosså et al., 2005).

3 Functional diversity refers to the characteristics of ecosystems and includes ecosystem complexity at different levels of organisation such as trophic levels Barbier, E. B., 2000a. Valuing the environment as input: review of applications to mangrove-fishery linkages. Ecological Economics, 35(1), 47-61, Cousins, S.H., 1991. Species diversity measurement:choosing the right index. Tree, 6, 190-192.. This approach uses trophic-level analysis to relate species diversity to functional ecosystem parameters such as food web structure or the transfer of energy, water and chemicals between different trophic levels. Functional diversity can be interpreted as the number of species required for a given ecological process.

Aggregations of orange roughy are also found in deep water coral environments (Koslow, 2003). Demersal species such as ling and tusk appear to be more common around corals than on the surrounding seabed (Costello et al., 2005; Husebø et al., 2002). The ivory tree coral, *Oculina varicosa*, located off the coast of Florida was found to be associated with grouper, snapper and amberjack (Reed, 2002) and Koenig (2001) observed a relationship between grouper, snapper, sea bass, and amberjack and the health (dead, sparse and intact) of Oculina colonies. Oculina reefs off Florida have been identified as essential fish habitat for federally-managed species, as have gorgonian-dominated deep coral communities off Alaska. Studies by Fosså et al. (2005; 2002) and Husebø et al. (2002) found that there was a greater abundance of species in coral areas than in non-coral areas. Husebø et al. (2002) noted that redfish, ling and tusk were larger and more abundant around coral habitats than non coral areas. There are possibilities that coral grounds act as spawning grounds and nurseries to juvenile fish; this evidence is however inconclusive (Costello et al., 2005; Fosså et al., 2002).

Much attention has recently been given to what has been coined essential fish habitat (EFH) (Anon., 1996; Peterson et al., 2000; Rosenberg et al., 2000). EFH is defined as “those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity” (Anon., 1996). Though definitive results are not yet available the research so far indicates that fish species exhibit facultative habitat use of cold-water coral (Auster, 2005). Facultative habitat use is defined as the use of habitat for many important life processes, but that the absence of these habitats does not result in extinction of the species in question. Hence, coral may not be essential habitat, but there seem to be indications that it may be a preferred habitat for many life processes, which infers that the destruction of such preferred habitat may result in losses connected with having to settle for second-best, reducing the value of the ecological good supplied.

There is limited research on the resilience of cold-water corals, but based on natural science evidence (Hughes and Stachowicz, 2004; Hughes et al., 2005; Reusch et al., 2005; Steneck et al., 2002; Steneck et al., 2004; Tilman et al., 2006) it is reasonable to assume that high levels of biodiversity in CWC ecosystems will contribute to higher resilience. Resilience is the capacity of an ecosystem to adapt to an external shock either by recovering to its original state or settling at a new equilibrium level. Ecosystems with higher biodiversity are thought to exhibit high resilience i.e. are better able to withstand unpredictable change than less diverse ecosystems (Holling, 1973; Holling et al., 1995). Marine ecosystems are also thought to have a higher level of resilience than terrestrial ecosystems (Holling et al., 1995). Maintaining a variety of organisms with different functional roles can ensure a greater variety of management options are available (Hooper et al., 2005).

3.2.2 Information Services

Cold-water corals are potential archives recording intermediate to sub-surface water temperatures and salinity (Lutringer et al., 2005). Recent concerns about climate change have emphasised the need for long-term proxies of climate change in the oceans (Risk et al., 2005). Cold-water corals may provide a unique record of temperature changes and serve as a good climate change proxy (Puglise et al., 2005).

Suggestions of a linkage between some cold-water corals and hydrocarbon seeps may lead to the use of corals as an indicator for oil and gas exploration (Rogers, 1999).

3.2.3 Biogeochemical Service

The most significant contributor to global warming is anthropogenic carbon dioxide (CO_2). The rapid release of CO_2 poses a fundamental threat to CWC ecosystems (Turley et al., 2007). The ever increasing search for new hydrocarbon resources and the emerging potential for CO_2 storage in the deep sea may have significant effects on deep-sea communities (Davies et al., 2007). The sequestration, that is the storage or absorption of CO_2, naturally or by human intervention, is high on the international climate policy agenda. Cold-water corals may sequester CO_2 and thus delay the release of CO_2 into the atmosphere. In this light, national policy for the protection of CWC could also potentially be used in a similar vein to afforestation thereby alleviating climate change. CWC ecosystems around the world are undergoing profound change due to economic development and it is to this issue which we now turn.

4. Human induced disturbances

Threats to corals include development activities such as fishing, hydrocarbon exploration, mineral mining and cable and pipe placement (Freiwald et al., 2004; Rogers, 1999). These activities damage CWC ecosystems and indirectly impose negative external costs, social costs borne by society. Significantly, each of these different activities imposes different levels of social costs. It is therefore important to consider these in turn. First we consider the effects of fishing.

By far the most serious anthropogenic threat to CWC reefs has so far been deep water trawling. Deep water trawling has two effects. First, harvesting of fish may have a direct impact on marine ecosystems because it affects the lower end of the food chain. Pauly et al. (1998) suggest that by fishing down the marine food web, ecosystem diversity is threatened and the system's resilience is weakened.

Second, there is evidence to indicate that trawling can have a direct physical impact on marine habitats and biodiversity. Watling and Langton (1994) reported that in untrawled areas, diversity of epibenthic invertebrates as well as diversity and abundance of young fish was greater than in trawled areas.

The physical effects of bottom trawling have been described as the marine equivalent to forest clear cutting acting as a major threat to biological diversity and economic sustainability (Watling and Norse, 1998). Trawl damage has occurred on many coral grounds, including those found in Norway, Ireland, Scotland, Alaska and Canada (Butler and Gass, 2001;Isså et al., 2002; Hall-Spencer et al., 2002; Krieger, 2001; Witherell and Coon, 2001). Fosså et al. (2002) estimated that 30 – 50% of CWC reefs in Norwegian waters have been damaged by trawl gear. Trawling can damage corals not only by crushing them but also by the suspension of sediments (Rogers, 1999).

Two further points are worth mentioning with respect to the effects of bottom trawling. First, the fragile physical features of CWC imply that it takes only a few sweeps by a bottom trawler to permanently damage coral structures. The impact of repeated trawls in a given area will impose smaller levels of damage. The resulting total damage function increases at an increasing rate, and the marginal damage function decreases at a decreasing rate with increasing fishing effort. Low levels of effort are considered to have a very high marginal impact because CWC areas are being fished by trawl gear for the first time. Once an area has been fished, subsequent effort has a smaller marginal impact because the most sensitive features of the habitat have already been destroyed.

The damage function has important economic consequences given the fact that CWC ecosystems may take many decades or centuries to recover from bottom trawling. It suggests that an optimal strategy may be to allow trawling in CWC areas that have already been heavily fished and are unlikely to recover but at the same time prevent bottom trawling in CWC areas that have not been affected by trawling. It is reasonable to assume that the functional values associated with undamaged CWC ecosystems will exceed the functional values of heavily fished areas and consequently these undamaged areas should receive a greater level of protection.

Secondly, potential for habitat damage by fishing gear in general depends on habitat characteristics (structural complexity, natural disturbance regimes), the size and type of fishing gear used, degree of contact with the sea bed, frequency of contact and the level of fishing effort (Jones, 1992). The development of a CWC habitat classification scheme that can accommodate an assessment of vulnerability to disturbance by different gear types is essential. Research on CWC ecosystems indicates that bottom trawl gears have the greatest impact on complex biogenic habitats (Anon., 2005). Observer data indicates that the rate of coral and sponge bycatch is about four times greater for bottom trawls than for longline or pot gear types for Alaska (Shester and Ayers, 2005). Table 1, developed from a review of the CWC literature, indicates different gear types and shows the level of severity due to different gears on the biological and physical structure of CWC. The physical structure represents damage to the reef structure and coral polyps, while the biological structure accounts for the impact of gear on the abundance and diversity of associated fauna surrounding CWC. Bottom trawling occurs throughout the world and there are many international examples of which we cite five (Ireland, Norway, U.S., Canada, Australia). Typically bottom trawling removes, displaces or physically breaks up corals which can end up in fishing nets. Bottom trawling is also thought to reduce habitat complexity and species composition in all five countries. Bottom trawling and dredging have the greatest impact because the area of seafloor is large, the forces on the seafloor are substantial and the spatial distribution of bottom trawling is extensive. Disturbance to CWC communities from other gear types is less harmful than mobile bottom – tending gear in all five countries.

A “risk evaluation” of this type could assist marine resource managers to assess the increasing gradation of risk attributable to different gear types in various habitats. This is important in conveying potential fishing impacts (and externalities) and options to marine resource managers. Table 1 shows that there is a general consensus across all five countries in terms of the level of damage caused by fishing gears on CWC, with bottom trawling and dredging having the highest impact on both physical and biological structures. It suggests that certain types of fishing could be permitted on CWC.

Gear type	Impact	Ireland	Norway	USA	Canada	Australia (Tasmania)
Long-lines (bottom set)	Physical Structure	2	2	1	2	-
	Biological Structure	1	1	1	1	-
Gill nets (bottom set)	Physical Structure	2	2	2	2	-
	Biological Structure	1	1	1	1	-
Bottom trawl	Physical Structure	3	3	3	3	3
	Biological Structure	3	3	3	3	3
Pots	Physical Structure	1	-	2	1	-
	Biological Structure	1	-	1	1	-
Dredge	Physical Structure	3	-	3	-	-
	Biological Structure	3	-	3	-	-

Table 1: The impact of different gear types on CWC based on current literature from five selected countries (Butler and Gass, 2001; Chuenpagdee et al., 2003; Fosså et al., 2002; Gass and Willison, 2005; Grehan et al., 2005; Koslow et al., 2001; Lumsden et al., 2007; Mortensen et al., 2005). The potential impacts on both the biological and physical structure of CWC are scored separately on a scale of 1 (lowest impact) to 3 (highest impact).

Other threats to CWC include hydrocarbon exploration, mineral mining and CO_2 pollution. Healthy CWC reefs are likely to be resilient to natural shocks such as adverse weather, geological phenomena (e.g. slides), changes in predator/prey relationships, and changes in abundance of species that support the reef's ecosystem. However, global pollution such as increased CO_2 and disturbance by mining and oil and gas exploration may weaken CWCs natural resilience to such shocks. Direct human impacts can also cause stress to CWCs. Oil and gas exploration takes place in many areas where cold-water corals occur; Gulf of Mexico, in the Campos Basin off Southeast Brazil, the UK, Norway and Mauritania (Freiwald et al., 2004). Such exploration can damage corals through the placement of rigs and pipelines in areas where corals are present and via the discharge of drill cuttings and suspension of sediments (Butler and Gass, 2001; Colman et al., 2005; Freiwald et al., 2004; Miller, 2001; Rogers, 1999). Damage can also be caused to nearby corals from the suspension of sediments in the placement of pipes and cables and from dumping of materials (such as ropes, nets and dredged sediments) on coral reef ecosystems (Freiwald et al., 2004).

5. CWC and their associated biodiversity: economic issues

There are a number of reasons why CWC ecosystems are threatened by human induced disturbance. CWC ecosystems in the high seas and within many Exclusive Economic Zones (EEZs) lack strong property rights, involve difficulties of excluding users and entail high monitoring and enforcement costs (Koenig, 2001; Long and Grehan, 2002). The lack of strong and complete property rights is a major cause of external costs and threats to CWC. Many of the goods and services provided by CWC ecosystems have public good attributes. Markets may fail to account for the value of CWC to society due in part to the public goods[4] nature of CWC and failure of markets to capture non-market benefits associated with the ecosystem.

Market failure also implies that resource users do not pay the full (external[5]) costs of any damage to CWC resulting from their actions[6]. Externalities affect a given marine ecosystem in various ways. Figure 1 indicates how type 2 conflicts result in over-harvesting of CWC fish species and impose higher external costs on other fishers.

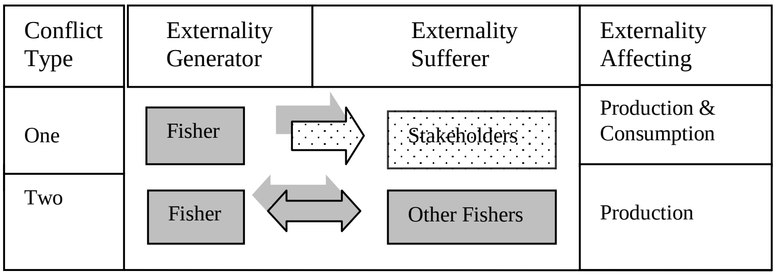

Figure 1: Examples of two types of conflict and associated external costs imposed on resource users (here exemplified by fishers and stakeholders) in the exploitation of CWC ecosystems.

These effects include possible damage to CWC habitat, reduction in fish stock and increased effort and costs of fish harvest. Type 1 conflicts, whereby fishers damage coral habitat, involves an asymmetrical externality arising in production.

The above mentioned social costs are borne by consumers, other stakeholders, the nation state and future generations. Because many of the goods and services including the genetic, functional and existence value associated with CWC are *unpriced*, fishers have little incentive to take account of these benefits in decision making. Figure 2 indicates three hypothetical equilibria for CWC ecosystems in relation to economic activity (Q) such as fishing effort. Under well defined property rights when external costs are not taken into account, private users will operate at Q^{Π} where marginal costs (MC) equal marginal benefits (MB). Under open access private users will operate at Q^{OA} where MB = AC

4 Indeed, fishers may, due to mismanagement of fish resources not even have the incentive to incorporate the ecosystem values of CWC for the relevant fishery as a whole. For example, since fisheries are often characterised by the so called "race for fish" where a single fisher has no incentive to take into account the effect of own harvesting upon other fishers, or even own well-being in the future, the fisheries benefits from the presence of CWC will face the same fate Clark, C. W., 2006, The Worldwide Crisis in Fisheries. Economic Models and Human Behavior. Cambridge University Press, Cambridge, UK.. Hence not only do fishers not take into account the benefits of CWC to people outside the fishery, the fisheries benefits are not taken into account either.

5 An absence of such external effects is one of the necessary conditions for market efficiency. Typically, the reason these benefits remain external to the market system is that they have the characteristics of public goods; in particular they are indivisible and perhaps also non-excludable, making their exchange in markets unlikely.

6 If an individual can benefit from a good irrespective of their contribution to it, or can avoid paying some of the costs inflicted on others, there is a tendency in both cases to maximise short-term self interest and to free-ride.

(Average Costs). Internalising external effects to achieve a social optimum (Q^S) results in an effort level that equates marginal external costs (MEC) with MB at a lower level of effort.

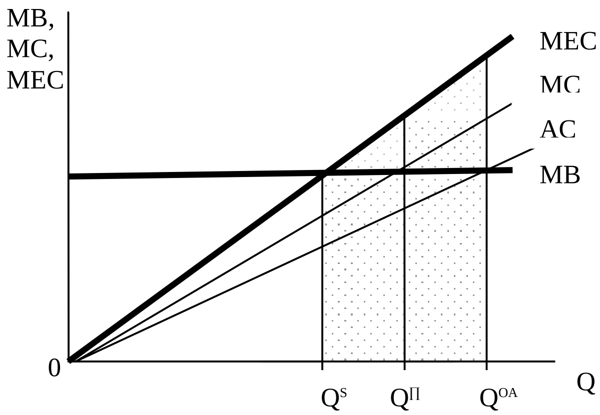

Figure 2: Optimal level of economic activity (Q) in a CWC ecosystem. The figure indicates open access (Q^{OA}) as well as private (Q^{Π}) and social equilibria (Q^S). The shaded area shows pareto relevant externality.

Economic theory suggests that internalising external costs will result in a level of economic activity that is socially optimal. This requires that external costs associated with development are measured in some way in order to derive an MEC curve. Valuation can be used to do this and can assist policy makers in developing regulatory measures to correct for pareto relevant external costs (shaded area shown in Figure 2).

5.1 Valuing CWC and its biodiversity

Assigning monetary values to non-market goods can potentially avoid undervaluation of CWC (Chee, 2004; Daily, 1997; Loomis et al., 2000). As shown in Figure 3, the concept of "Total Economic Value" (TEV) can be used to describe the components of value associated with a CWC ecosystem. From an economic perspective, the goods and services generated by a CWC ecosystem comprise of *use* and *non-use* values.

Direct use values refer to the actual and/or planned use of a service by an individual (Bolt et al., 2005). Direct use values are based on stocks or assets, such as coral and fish biomass and genetic raw materials embodied in organisms which are sold in markets and services derived from a CWC area. These goods and services are traded and they include the value of contracts for commercial seafood products, coral for jewellery, and organisms and raw materials extracted from CWC sites which are used as inputs by manufacturing and pharmaceutical industries. Market analysis is used to estimate their value (Figure 3).

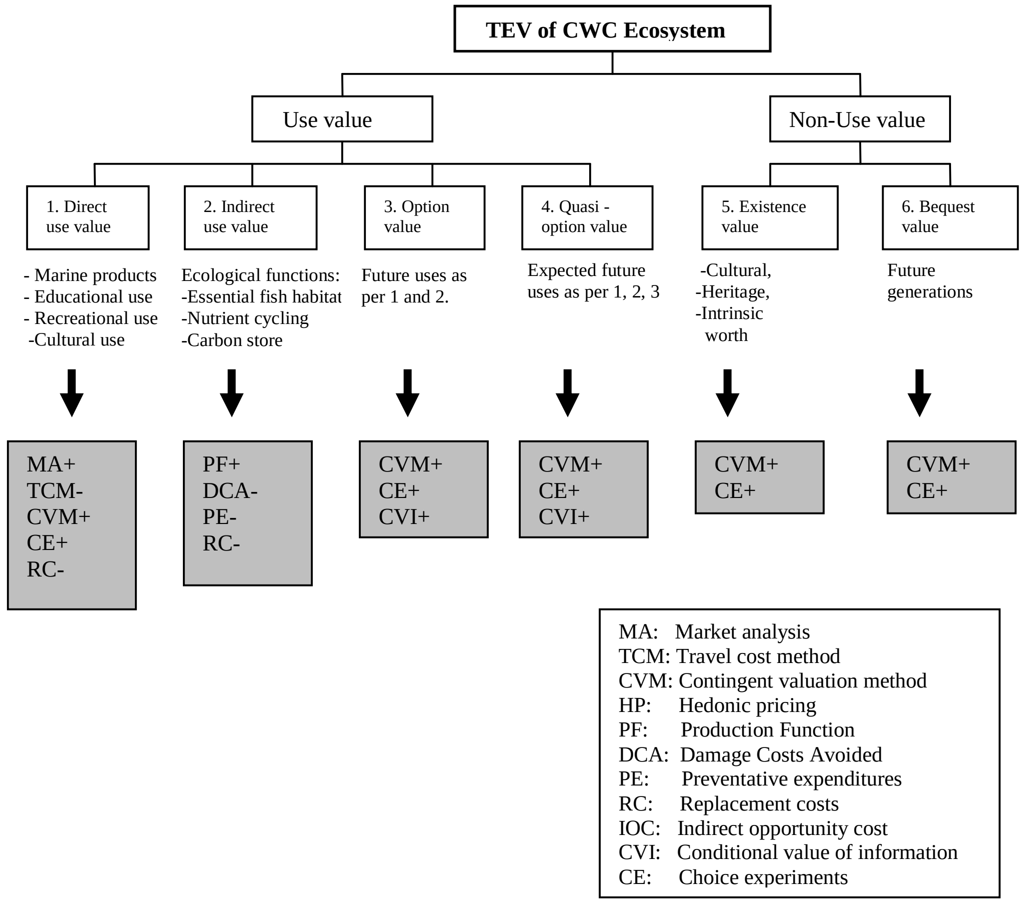

Figure 3: Components of Total Economic Value associated with CWC ecosystems. The + indicate high relevance and – indicate low relevance in the use of the valuation technique for the good or service in question. Source: adapted from Barbier et al (1994).

The valuation of CWC goods and services which are not traded is more problematic. In the absence of market prices for CWC values certain non-market valuation techniques need to be used. Two broad approaches are relevant to the valuation of CWC ecosystems: surrogate market valuation and stated preference methods. Surrogate market valuation methods of relevance include the travel cost method (TCM), and the production function (PF) approach. Services provided by CWC include non-market use values such as scientific, aesthetic and educational information and potential recreational viewing of reefs. The contingent valuation method (CVM) could be employed to estimate the aesthetic and scientific value of CWC. TCM has been widely used for valuing the non-market benefits associated with visiting marine parks, protected areas and dive sites (Bhat, 2003; Carr and Mendelsohn, 2003; Soderqvist et al., 2005). Although tours in deep water submersibles are planned for the Titanic and would also be feasible for cold-water corals (Ecoserve, 1998), the application of TCM to CWC at this time is likely to be limited because so few if any CWC sites can be viewed directly on any scale by the public.

Indirect use values include nutrient cycling, carbon sequestration, nursery grounds and preferred habitat for commercial fish species. The production function approach represents an important means of quantifying functional values associated with CWC. The method would link CWC reefs to fisheries, identifying to what degree profits from commercial species are affected by the presence or absence of CWC. Given the identification of such a link, this could then be modelled in order to ascertain the losses involved when this link is not included in management or conservation decisions. The method can be used to take account of how changes in habitat area or quality affects production (Barbier, 2000b; Knowler, 2002). A number of studies from other ecosystems have been conducted using this approach to determine the indirect value of mangroves and marshlands as inputs in fishery production (Barbier, 1994, 2000b; Barbier and Strand, 1998; Bell, 1997; Daily, 1997; Ellis and Fisher, 1987; Sathirathai and Barbier, 2001). Barbier and Strand (1998) established a value for one of the non-market functions of mangroves by exploring the relationship between mangroves and shrimp production in Campeche, Mexico. In a similar study, Barbier et al (2002) developed a dynamic production function approach to analyze the influence of habitat changes on marine demersal and shell fisheries in Thailand. Other studies have looked at the value of marshlands for Gulf Coast fisheries in the southern United States (Bell, 1989, 1997; Ellis and Fisher, 1987; Lynne et al., 1981). Anderson (1989) developed a simple model to generate approximate estimates of some of the economic benefits that would accrue from sea grass restoration which serve as a preferred habitat for the blue crab.

Although there are problems associated with double counting (Aylward and Barbier, 1992), the production function approach is highly relevant to CWC in view of the functional values of the ecosystem as discussed above. Indeed these functional values probably represent the strongest single economic argument for their preservation since they may support an extractive fishing industry which generates several hundreds of millions of euros per year, employing millions of fishers globally.

Use values also include option value, i.e. the value of the option to guarantee use of the service by the individual in the future (Spaninks and van Beukering, 1997; Weisbrod, 1964). Bishop (1982) revealed that where an individual is certain of demand but faces uncertain supply of a good they would be willing to pay an option price to avoid the future risk of losing the environmental good. Freeman (1985) demonstrated that this does not hold for all forms of uncertainty of supply and showed that supply-side option value could be either positive or negative.

Our observations and field research indicate that quasi option values (QOV) constitute a particularly important set of values for CWC. In circumstances when policy makers are not certain about the size of the benefits or costs of alternative uses of a resource, decision making should be modified to reflect this uncertainty. Arrow and Fisher (1974) used the term to describe the welfare gain associated with delaying a decision when uncertainty exists about the payoffs of alternative choices and when the choices involve an irreversible commitment of resources. A number of more recent studies indicate that QOV refers to the value of future information protected by delaying an irreversible development (Arrow and Fisher, 1974; Perman et al., 2003). Although the benefits from preserving CWC habitats are uncertain at this time, these benefits could become more certain through time as information increases about the uses to which the coral habitats and their species can be put. If development of activities such as fishing or mining proceeds, this source of functional and genetic information and their associated services are lost forever. There is scientific evidence to indicate that CWCs are being destroyed quite rapidly (Fosså et al., 2002). This is of some concern in view of the fact that CWCs generally do not recover from this damage, that is, the damage is irreversible. Development activities such as fishing may also reduce the amount of information available to scientists at CWC sites. When scientists discover new coral areas they frequently observe that they are damaged by developments (trawling etc.) before they are even discovered and recorded, and before scientists have had a chance to learn more about their uses.

The literature suggests that if the expected growth of information is independent of developments, i.e. we do not need developments to generate information, then QOV will be positive (Freeman, 1984). The main threat to CWC ecosystems is bottom trawling. Arguably, bottom trawling does not generate new information and therefore QOV is likely to be positive.

A number of other empirical studies place emphasis on the importance of QOV in forests (Albers, 1996; Albers and Robinson, 2007; Bulte et al., 2002; Simpson et al., 1996) and protected areas (Cullen, 1994). Option value and quasi-option value is of greater significance to users than non-users (Walsh et al., 1984). The main beneficiaries (of option and quasi-option values) are likely to be resource users - fishing communities and to a lesser extent the marine scientific community involved in medicinal and pharmaceutical research. Our own observations suggest that it is clear from campaigns led by coastal fishers (who do not bottom trawl) against other fishers (who do bottom trawl), that deep-water corals may have significant quasi option values to inshore fishermen (Armstrong and van de Hove, 2008). Coastal fishers are seen to bear the external costs generated by Norwegian trawlers as indicated in Figure 2. CVM represents a useful technique to estimate option and QOV with respect to CWC.

More recently, empirical studies on marine values have placed emphasis on non-use value (right hand side of Figure 3). These refer to situations where an individual knows a biological resource exists and will continue to exist, independently of any actual or prospective use by the individual and where that individual would feel a 'loss' if the resource were to disappear (Brown, 1990; Ledoux et al., 2001). Non-use values include the following: *bequest value*: the value of ensuring that the resource remains intact for one's future heirs (Krutilla, 1967); *existence value:* the value that arises from ensuring the survival of a resource (Pearce and Turner, 1990; Perman et al., 2003).

The contingent valuation method (CVM) and choice experiments are tried and tested techniques capable of measuring non-use values associated with CWC. These methods can be used to forecast preferences for CWC habitat and biodiversity or to determine if consumers are willing to pay a price premium for fish caught in CWC areas using sustainable fishing methods. For example a number of studies indicate that consumers prefer ecolabeled products, as long as price premiums for the ecolabeled products are not large (Donath et al., 2000; Gudmundsson and Wessells, 2000; Johnston et al., 2001; Wessells et al., 1999). The use of less damaging gear could lead to less impact to CWC but also allow fishing to continue thereby protecting fishing communities and their livelihoods. However, it has been argued that lack of familiarity with a good causes invalid responses in stated preference surveys (Cameron and Englin, 1997; Whitehead et al., 1995) (although see Turpie (2003) and Kniivilä (2006) for a counter point of view). This issue may be important in view of the fact that the public know very little about CWC ecosystems. Where the public are not familiar with a good, Spash and Hanley (1995) and Christie et al. (2006) emphasise the importance of information, learning and focus groups as part of the valuation process in order to improve the precision of willingness to pay (WTP) estimates (Aldred and Jacobs, 2000; Alvarez-Farizo and Hanley, 2006; Kenyon and Nevin, 2001). Provided the issue of lack of familiarity with CWC can be overcome, stated preference techniques could provide critical information to policymakers and producers about a range of potential future values associated with CWC.

The main beneficiaries of existence values are probably the general public. A number of international environmental NGOs have been particularly vocal in pressing for CWC conservation, for example WWF, Oceana and UNEP.

	Value		
	Use		Non-use
	Direct	Indirect	
Goods/assets			
Medical/genetic resources	○○○		
Raw materials	●●		
Ornamental resources	●●		
Services			
Nutrient retention		○	
Carbon store		○○	
Habitat functions		●●	
Refugium function		●●	
Nursery function		●●	
Aesthetic information	○		
Recreation	○		
Science & education information	○		
Attributes/diversity			
Biological diversity			●●●
Cultural heritage uniqueness			○○
Intrinsic worth			○○

● = economic values available in peer-reviewed literature; ● = low; ●● = medium; ●●● = high;
○ = no economic values available in peer reviewed literature, but economic values probable; ○ = low; ○○ = medium; ○○○ = high.

Table 2: Cold-water coral in the benthic zone, values of goods and services.

Although this has not been quantified, this provides some evidence of the importance of non-use values in relation to CWC. It is clear from NGO involvement that deep-water corals have both existence and bequest values. Table 2 qualitatively sums up the values of the goods and services in CWC, based on current availability.

5.3 Uncertainty and CWC ecosystems

Although valuation is useful in capturing the value of CWC ecological goods and services there are a number of problems with this approach. Research on biodiversity by Perrings and Pearce (1994) suggest there may be a threshold where resilience is threatened and the ecosystem is irreversibly changed. In circumstances where resilience is not threatened, organisms or physical characteristics of CWC ecosystems may have little or no (functional) economic value. The problem is fundamentally one of uncertain ecological thresholds which imply that damage functions are not smooth and continuous in relation to changes in economic activity but "jump" once economic activity exceeds a certain level (Q^S) whereby marine ecosystems are driven beyond some critical point (a) to a new point involving much higher external costs (b) as shown in Figure 4.

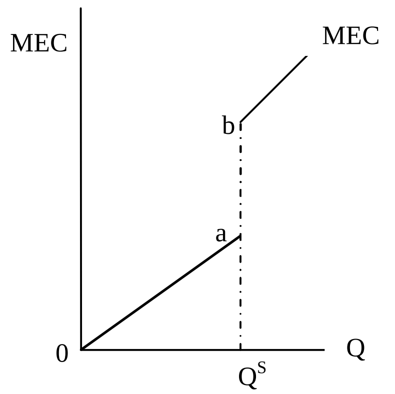

Figure 4: Q^S indicates a critical threshold which if crossed leads to a "jump in the marginal damage function between points a and b with an increase in economic activities (Q) such as fishing.

The existence of thresholds in CWC ecosystems remains uncertain in part because research on CWC is at a very early stage. Thresholds are known to occur in many marine habitats including tropical corals (Moberg and Folke, 1999) and might be expected to occur in CWC ecosystems.

It seems clear that there are externalities connected to CWC ecosystems. What instruments are relevant for dealing with these issues? Is there a role for property rights or other economic mechanisms? Economic theory indicates that providing an individual with a right to use a resource will reduce or eliminate the negative externalities caused for instance by overfishing or even the destruction of marine habitats. We now explore this issue.

6. CWC ecosystems and biodiversity: policy issues

Taxes or transferable quotas have been suggested as modes of managing marine resources. Taxation of harvest or effort has been discussed (Jensen and Vestergaard, 2003), but never implemented other than in addition to setting total allowable harvests (Oelofsen, 1999). Transferable quotas are much used in fisheries management for harvest shares (Hannesson, 2004), but have not been applied with regard to harvest areas.

In theory, transferable quotas could be used to protect the functional values connected to different habitats (Holland and Schnier, 2006). However, transferable quotas are more suited to resources of a more homogenous and renewable character than CWC such as fishing and timber harvesting but much less suitable for resources that involve irreversible losses such as CWC or wetland ecosystems.

The near irreversible effects of trawling upon CWC clearly sets the scene for either 1) a precautionary approach[7] allied to changes in the use of fishing gear, or 2) some determination of the optimal level of CWC destruction. We discuss the need for a precautionary approach first.

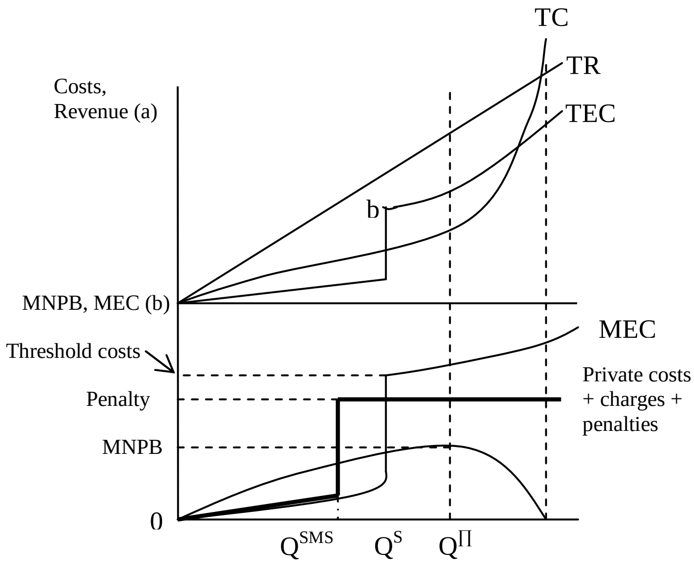

Figure 5: Penalty and threshold costs where thresholds are not known with certainty (Adapted from Perrings and Pearce (1994)). Slide (a) indicates Total costs (TC), Total revenue (TR) and shows a discontinuous Total External Cost (TEC) function. Slide (b) shows marginal external costs (MEC), marginal net private benefit (MNPB) and a private costs and charges plus penalties function.

Given the importance of threshold effects in marine ecosystems, there may be a role for standards. In many ecosystems thresholds are not known with certainty. Figure 5 (a) shows that if left unregulated, the external costs connected to a CWC resource can be expected to be discontinuous at some level of

7 The policy of taking action before uncertainty about possible environmental damages is resolved has been referred to as the 'precautionary principle'. One justification for this is that the costs of damage to biological resources may exceed the costs of preventative action Taylor, P., 1991. The precautionary principle and the prevention of pollution ECOS, 12(4), 41-46.. Also, irreversible damage may occur, such as species extinctions. The emphasis is thus on avoiding potentially damaging situations in the face of uncertainty over future outcomes Haigh, N., 1993, The precautionary principle and British environmental policy. Institute for European Environmental Policy, London, Myers, N., 1992. Population/environment linkages: discontinuities ahead. Ambio, 21, 116-118.. The precautionary principle could also be implemented by using marine reserves as a part of fisheries management Lauck, T., Clark, C. W. , Mangel, M. and Munro, G, 1998. Implementing the precautionary principle in fisheries management through marine reserves. Ecological Applications, 8(1), 72-78..

economic activity Q^S and may jump at this point to some much higher level, and then continue to increase with the level of development. If regulation is not introduced to protect the coral habitat, the private fisher will operate at Q^{Π} where Marginal Net Private Benefit (MNPB) is maximised. If the threshold values for CWC habitats and their biodiversity are not known with certainty then a safe minimum standard (SMS) may need to be set before the threshold (Ciracy-Wantrup, 1952). In Figure 5(b) the threshold costs (the social costs borne by society) associated with a level of economic activity that is greater than Q^S clearly exceeds marginal net private benefit and therefore requires regulation. In the regulated case a level of economic activity is set such that exceeding the safe minimum standard (Q^{SMS}) incurs a penalty plus charges (black line) in excess of MNPB. This will result in a social optimum (Q^{SMS}).

The existence of threshold effects invalidates normal tests for the efficiency in the allocation of resources. Discontinuities imply that it is no longer feasible to equate marginal net private benefit with marginal external cost. The implication being that regulatory measures to protect certain thresholds cannot rely solely on economic criteria. They are set according to ethical judgements about some socially acceptable margin of safety. The precautionary principle is seen as an ethical judgment about how society should respond to risks in the face of collective ignorance about the future impacts of development activities on the environment (Perrings and Pearce, 1994).

We think there is probably a good case for using the precautionary approach in circumstances where it is thought that functional values are high. Other approaches should be considered where this is not the case. These alternatives need to be considered in the context of provision of an “optimal” level of CWC habitat. The question being; have we already destroyed as much as or more than the optimal amount of CWC habitats, or can we safely allow more destructive behaviour? Clearly when CWC was initially being investigated in the late 1990s the scarcity value was high as there were very few known CWC areas. The scarcity value has been declining as more and more CWC is being discovered (Armstrong and van de Hove, 2008). Part of the answer to this question depends on the type of values associated with a given CWC ecosystem. If the external costs of development are composed mainly of existence values then it may not be vital to protect CWC in every locality. If instead CWC ecosystem goods and services are made up mainly of functional values then the scarcity value is less relevant. In this case development will undermine economic activity and it will be important to protect coral habitat in each locality because they indirectly support economic activities such as commercial fishing wherever they occur. The increasing prevalence of CWC begs the question as to whether it is of greater importance to ocean life than previously thought. An important policy question which follows from this is: does this entail a more active approach to protection and conservation using for instance a precautionary approach?

We think protection and conservation is warranted under conditions that involve a high level of uncertainty. Arguably there is a good case to be made for the use of the precautionary approach (i.e. the designation of areas where coral distribution is high such as marine protected areas (MPA)) under circumstances where external costs of development are likely to be high. An MPA refers to a management area in which usage is regulated by zoning for different activities (Sumaila et al., 2000). MPAs can be applied at different levels including complete closures, gear restrictions or zoning approaches in which different activities are permitted or banned in particular zones. Broadly speaking, high uncertainty coral habitats could be distinguished from low uncertainty coral habitats. The former would have the following characteristics: where CWC ecosystems involve high levels of biodiversity, functional values and the existence of threshold effects and irreversibilities; where scientists have discovered or suspect newly recorded CWC ecosystems that are not damaged by developments and where there is evidence to suggest that CWC habitats are especially sensitive to disturbance. In such areas conservation MPAs could be introduced that are designed to protect fish stocks as well as coral habitat. MPAs should also be used where effective enforcement or compliance costs (in order to protect CWC) are known to be very high and exceed the value of development. A special case of introducing temporary MPAs may need to be made in circumstances where scientists suspect (but have not demonstrated) that high uncertainty CWC ecosystems occur. This requires dynamic MPAs that are flexible and can be quickly opened up to development in circumstances where CWC sites are not deemed to be worthy of protection. This approach may be particularly relevant in safeguarding option and quasi option values.

MPAs involve significant costs in their management but also by restricting economic activities. Under circumstances which involve a low degree of uncertainty, alternative regulatory mechanisms to MPAs may be more appropriate. These include “low uncertainty” areas that can be categorised as follows: heavily fished areas where habitat damage has already occurred, where functional values are low; where benefits of fishing or other developments are high and damage to CWC is low; where fishers can demonstrate that economic activity is not damaging to CWC habitat.

Several countries appear to show increasing interest in establishing MPAs based on the precautionary principle in order to hedge against our ignorance of CWC (Bohnsack, 1998). In 2000 the South Atlantic Fishery Management Council established an *Oculina* Habitat Area of Particular Concern (OHAPC) along the eastern Florida shelf that prohibited trawls, dredges, traps, and long lines in an area covering 1029 KM^2 (Reed, 2002). Coral sites in the Gulf of Alaska, Aleutian Islands, Florida and the Gulf of Mexico have been recognised as Habitats of Particular Concern and designated as MPAs under the Magnuson Stevens Act (Lumsden et al., 2007).

Ireland, Sweden and the UK, have designated sites as Special Areas of Conservation (SACs) under the EU Habitats Directive. The Darwin mounds, lying off the North West coast of Scotland, were the first offshore areas to be protected with an emergency closure under the European Common Fisheries Policy in 2003 (Davies et al., 2007). In 1999, the Australian government declared the Tasman Seamounts Reserve under the National Parks and Wildlife Conservation Act 1975. The management plan came into effect in 2002 under the Biodiversity and Conservation Act 1999 (Freiwald et al., 2004).

The Norwegian Ministry of Fisheries passed legislation in 1999 to introduce partial closures to protect reefs and the ocean floor from the impacts of fishing (Fosså et al., 2005). Five Norwegian reefs have been closed to trawling, including the Røst reef, the world's largest known *Lophelia* reef (Fosså et al., 2005; Fosså et al., 2002). In 2002, Canada divided 424km^2 in the Northeast Channel area into two zones with 90% closed to all bottom fishing and allowed longlining in the remaining 10% under the Fisheries Act (Butler, 2005). New Zealand has also acted to protect 19 seamounts from bottom trawling (Freiwald et al., 2004).

Political acceptance of MPAs may be problematic. Frequently they imply foregoing economic development in order to avoid an event that might happen. Such arguments are generally not popular with fishers. Also MPAs may not guarantee that CWC areas are not damaged by fishing. Koenig (2001) reports that 90% of the Oculina habitat within the experimental Oculina Research Reserve was destroyed due to poaching by illegal trawlers between 1970 and 2001. Where the cost of excluding fishers is high the best option may be to prevent fishing altogether and impose very stiff penalties for non compliance.

Scientists are suggesting compromises involving limited development options which maintain the integrity of CWC ecosystems but allow development. Shester and Ayers (2005) show how a cost effectiveness approach can be used to allow bottom trawling in Alaska only in areas of high fish harvest and low habitat impact.

Crucially, well designed policy instruments must include strong stewardship incentives which ensure that security of tenure is conditional upon management systems that protect CWC habitat. Property rights systems need to nudge the fishing sector toward adopting management methods that conserve and safeguard CWC ecosystems. Management systems that make fishers a stakeholder in the ecosystem, with an interest in the future health and management of the ecosystem provide the right incentives (Arnason, 1990, 1998). Harvesting rights are devalued if essential fish habitat is damaged through overharvesting and other developments. This shifts the "burden of proof" from the regulator to fishers who must demonstrate responsible management of the wider ecosystems in order to continue fishing (Charles, 2002; Sanchirico, 2008; Young, 1995, 1999). If fishers with such rights protect CWC habitats their activities will yield positive externalities: protecting functional, heritage and existence values of CWC ecosystems thereby contributing to the management of an important marine public good.

7. Conclusions

Although scientific knowledge of cold-water corals is still at a very early stage, a clear message which arises from this study is that CWC ecosystems are not valueless. Whilst a clear empirical link between CWC habitat and commercial fish species has not been shown, studies to date indicate that functional values in support of commercial fisheries probably represent the most important service provided by CWC. Increasing public support for NGOs campaigning to conserve CWC would also appear to indicate that existence values are not insignificant. Viewing values associated with CWC habitats are of minor importance at this time. The economic value of CWC species in formulating pharmaceutical compounds is thought to be promising but clinical trials are still at a very early stage.

Although bottom trawling is not a problem in all CWC habitats, bottom fishing appears to constitute the most serious single threat to CWC goods and services by far. This is of some concern in view of the fact that CWCs generally do not recover from this damage. Much CWC degradation is therefore irreversible. It is also thought that the most sensitive features of CWC habitats are destroyed after only a few sweeps by a trawler. Trawling may involve a decline in economic values associated with fishing, medicinal values, the value of CWC as a carbon sink and existence values. It is difficult to quantify these external costs but it is certainly clear from initial research using stated preference methods, as well as NGO involvement and pressure from coastal fishing groups that deep-water corals may have significant quasi option values to inshore fishermen (Armstrong and van de Hove, 2008) as well as existence values for the general public.

MPAs should be implemented in undamaged CWC habitats in circumstances that involve high functional values, high uncertainty or the existence of threshold effects. Fishing activity in MPAs should be confined to vessels using gear types that do not threaten CWC. Well designed mechanisms should be introduced that shift the burden of proof from regulators to fishers. This should be underpinned by training programmes that convey information about the functional role of CWC ecosystems to fishers. In order to continue fishing, fishers need to be able to demonstrate responsible fishing. This will accord with the FAO Code of Conduct for Responsible Fisheries (Long, 2007). Steps need to be taken to reduce the costs of exclusion in both MPAs and in areas involving habitat ITQs in order to make exclusion more effective. Where the cost of exclusion is very high, trawling should not be allowed and in these areas penalties to poachers should be stiff enough to deter future illegal fishing.

In the absence of information on existence values, there is some evidence, to suggest that the main external costs are borne locally by other users (i.e. fishers) of the same ecosystem. Given this, and the fact that CWC are mainly threatened by trawling, this would tend to point toward a more active approach to protection at the local level. This makes the problem easier to address since it can be done at the national level and does not require negotiation and agreement between countries.

For the future, at least three avenues of research are worth pursuing. First, more research is required to weigh up the costs and benefits to the fishing sector on alternative gear types in different seabed CWC habitats that allow fishing but minimise coral damage. It is still not clear how much the deep sea fishing sector would stand to lose by adopting more coral friendly fishing practices. More detailed work along the lines carried out by Shester and Ayers (2005) is required to answer this question. Second, more information on the public good nature of CWC ecosystems would facilitate in the formulation of CWC policy in the high seas. Third, more research is required in clarifying the precise economic nature of linkages between the environmental regulatory function of CWC and the economic activity it supports (i.e. commercial fish production). This research should focus on identifying the particular characteristics of CWC ecosystems that can be used to evaluate the impact of disturbance on CWC ecosystem health, pointing towards precautionary measures in the face of possible option values.

References

Albers, H.J., 1996. Modelling ecological constraints on tropical forest management: spatial interdependence, irreversibility, and uncertainty Jounal of Environmental Economics and Management, 30(1), 73-94.

Albers, H.J. and Robinson, E.J.Z., 2007. Spatial-temporal aspects of cost benefit analysis for park management: an example from Khao Yai national park Thailand. Journal of Forest Economics, 13(2-3), 129-150.

Aldred, J. and Jacobs, M. , 2000. Citizens and wetlands: evaluating the Ely citizens' jury. Ecological Economics, 34(2), 217-232.

Alvarez-Farizo, B. and Hanley, N., 2006. Improving the process of valuing non-market benefits: combining citizens' juries with choice modelling. Land Economics, 82(3), 465-478.

Anderson, E.E., 1989. Economic benefits of habitat restoration: seagrass and the Virginia hard-shell blue crab fishery. North American Journal of Fisheries Management, 9(2), 140-149.

Anon., 1996, The Sustainable Fisheries Act of 1996. USA.

Anon., 2004. The stock book - annual review of fish stocks in 2004 with management advice for 2005, Marine Institute.

Anon., 2005. The stock book - annual review of fish stocks in 2005 with management advice for 2006, Marine Institute.

Armstrong, C. W. and van de Hove, S., 2008. The formation of policy for protection of cold-water coral off the coast of Norway. Marine Policy, 32(1), 66-73.

Arnason, R., 1990. Minimum information management in fisheries. Canadian Journal of Economics, 23, 630-653.

Arnason, R., 1998. Ecological fisheries management using individual transferable share quotas. Ecological Fisheries Management, 8(1), 151-159.

Arrow, K.J. and Fisher, K.C., 1974. Environmental preservation, uncertainty and irreversibility. Quarterly Journal of Economics, 88, 312-319.

Auster, P.J., 2005, Are deep water corals important habitats for fishes? In: A Freiwald and J. Roberts (Eds.), Cold-Water Corals and Ecosystems. Springer-Verlag, Berlin Heidelberg, pp. 747-760.

Aylward, B. and Barbier, E. B., 1992. Valuing environmental functions in developing countries. Biodiversity and Conservation, 1(1), 34-50.

Barbier, E. B., 1994. Valuing environmental functions: tropical wetlands. Land Economics, 70, 155-173.

Barbier, E. B., 2000a. Valuing the environment as input: review of applications to mangrove-fishery linkages. Ecological Economics, 35(1), 47-61.

Barbier, E. B., 2000b. Valuing the environment as input: review of the applications to mangrove-fishery linkages. Ecological Economics, 35, 47-61.

Barbier, E. B., Burgess, J.C. and Folke, C., 1994, Paradise lost? The ecological economics of biodiversity. Earthscan, London.

Barbier, E. B. and Strand, I., 1998. Valuing Mangrove-Fishery Linkages - A Case Study of Campeche, Mexico. Environmental and Resource Economics, 12, 151-166.

Barbier, E.B., Strand, I. and Sathirathai, S., 2002. Do open access conditions affect the valuation of an externality? Estimating the welfare effects of mangrove-fishery linkages in thailand. Environmental and Resource Economics, 21, 343-367.

Beaumont, N. J., Austen, M. C., Atkins, J. P., Burdon, D., Degraer, S. et al., 2007. Identification, definition and quantification of goods and services provided by marine biodiversity: Implications for the ecosystem approach. Marine Pollution Bulletin, 54(3), 253-265.

Bell, F.W., 1989. Application of the wetland valuation theory to Florida fisheries. 95, Florida State University, Tallahassee.

Bell, F.W., 1997. The economic value of saltwater marsh supporting marine recreational fishing in the southeastern United States. Ecological Economics, 21, 243-254.

Bhat, M., 2003. Application of non-market valuation to the Florida Keys marine reserve management. Journal of Environmental Management, 67(4), 315-325.

Bishop, R.C., 1982. Option value: an exposition and extension. Land Economics, 58(1), 1-15.

Bohnsack, J. A., 1998. Application of marine reserves to reef fisheries management. Austral Ecology, 23(3), 298-304.

Bolt, K., Ruta, G. and Sarraf, M., 2005, Estimating the cost of environmental degradation - a training manual. World Bank.

Brown, G.M., 1990, Valuation of genetic resources. In: G.H. Orians, G.M. Brown, W.E. Kunin and J.E. Swierbinski (Eds.), The preservation and valuation of biological resources. University of Washington Press, Seattle.

Bulte, E., van Soest, D.P., van Kooten, G.C. and Schipper, R.A., 2002. Forest conservation in Costa Rica when non-use benefits are uncertain but rising. American Journal of Agricultural Economics, 84(1), 150-160.

Butler, M., 2005, Conserving corals in Atlantic Canada: a historical perspective. In: A. Freiwald and J.M. Roberts (Eds.), Cold-water corals and ecosystems. Springer Verlag, Berlin Heidelberg, pp. 1199-1209.

Butler, M. and Gass, S., 2001, How to protect corals in Atlantic Canada. In: J.H.M. Willison, J. Hall, S. Gass, E. Kenchington, M. Butler et al. (Eds.), First International Symposium on Deep-Sea Corals. Ecology Action Centre, Halifax, Nova Scotia, pp. 156-165.

Cameron, T.A. and Englin, J., 1997. Respondent experience and contingent valuation of environmental goods. Journal of Environmental Economics and Management, 33, 296-313.

Carr, L. and Mendelsohn, R., 2003. Valuing coral reefs: a travel cost method of the Great Barrier Reef. Ambio, 32(5), 353-357.

Charles, A.T., 2002. The precautionary approach and 'burden of proof' challenges in fisheries management. Bulletin of Marine Science, 70(2), 683 - 694.

Chee, Y.E., 2004. An ecological perspective on the valuation of ecosystem services. Biological Conservation, 120(4), 549-565.

Christie, M. , Hanley, N., Warren, J., Murphy, K. , Wright, R. et al., 2006. Valuing the diversity of biodiversity. Ecological Economics, 58(2), 304-317.

Chuenpagdee, R., Morgan, L.E., Maxwell, S.M., Norse, E.A. and Pauly, D., 2003. Shifting gears: assessing collateral impacts of fishing methods in US waters. Front. Ecol. Environ, 1(10), 517-524.

Ciracy-Wantrup, S., 1952, Resource conservation: economics and policies. University of California Press, Berkeley.

Clark, C. W., 2006, The Worldwide Crisis in Fisheries. Economic Models and Human Behavior. Cambridge University Press, Cambridge, UK.

Colman, J.G., Gordon, D.M., Lane, A.P., Forde, M.J. and Fitzpatrick, J.J., 2005, Carbonate mounds off Mauritania, Northwest Africa: status of deep water corals and implications for management of fishing and oil exploration activities. In: A Freiwald and J. Roberts (Eds.), Cold-water corals and ecosystems. Springer-Verlag, Berlin Heidelberg, pp. 417-441.

Costello, M.J., McCrea, M., Freiwald, A., Lundalv, T., Jonsson, L. et al., 2005, Role of cold-water Lophelia pertusa coral reefs as fish habitat in the NE Atlantic. In: A. Freiwald and J.M. Roberts (Eds.), Cold-water corals and ecosystems. Springer-Verlag, Berlin Heidelberg, pp. 771-805.

Cousins, S.H., 1991. Species diversity measurement:choosing the right index. Tree, 6, 190-192.

Crozier, R.H., 1992. Genetic diversity and the agony of choice. Biological Conservation, 61(1), 11-15.

Cullen, R., 1994. Antarctic minerals and conservation. Ecological Economics, 10(2), 143-155.

Daily, G.C., 1997, Nature's Services: Societal Dependence on Natural Ecosystems Island Press, Washington DC.

Davies, A.J., Roberts, J.M. and Hall-Spencer, J., 2007. Preserving deep-sea natural heritage: emerging issues in offshore conservation and management. Biological Conservation, 138, 299-312.

Donath, H., Wessells, C.R., Johnston, R.J. and Asche, F., 2000, Consumer preferences for ecolabeled seafood in the United States and Norway: a comparison, IIFET, Oregon, USA.

Ecoserve, 1998, Coral reefs in irish waters.

Ehrlich, H., Etnoyer, P., Litvinov, S.D., Olennikova, M.M., Domaschke, H. et al., 2006. Biomaterial structure in deep-sea bamboo coral (Anthozoa: Gorgonacea: Isididae): perspectives for the development of bone implants and templates for tissue engineering. Materialwissenschaft und Werkstofftechnik 37(6), 552-557.

Ellis, G.M. and Fisher, A.C., 1987. Valuing the environment as an input. Journal of Environmental Management, 25, 149-156.

Fosså, J.H., Lindberg, B., Christensen, O., Lundalv, T., Svellingen, I. et al., 2005. Mapping of *Lophelia* reefs in Norway: experiences and survey methods. In: Freiwald, A, and Roberts JM (eds), 2005, *Cold water corals and ecosystems.* Springer verlag Berlin Heidleberg, 359-391.

Fosså, JH, Mortensen, PB and Furevik, DM, 2002. The deep water coral *Lophelia pertusa* in Norwegian waters: distribution and fishery impacts. . Hydrobiologia, 471, 1-12.
Freeman, A. M., 1985. Supply uncertainty, option price and option value. Land Economics, 61(2), 176-181.
Freeman, A. M. III, 1984. The quasi-option value of irreversible development. Journal of Environmental Economics and Management, 11(3), 292-295.
Freiwald, A., Fosså, J.H., Grehan, A., Koslow, T. and Roberts, J.M. , 2004. Cold-water Coral Reefs, UNEP-WCMC, Cambridge, UK.
Gass, S. E. and Willison, J.H.M., 2005, An assessment of the distribution of deep-sea corals in Atlantic Canada by using both scientific and local forms of knowledge In: A Freiwald and JM Roberts (Eds.), Cold-water corals and ecosystems. Springer Verlag, Berlin Heidelberg, pp. 223 - 245.
Grehan, A. J., Unnithan, V., Olu, K. and Opderbecke, J., 2005, Fishing impacts on Irish deep-water coral reefs: making the case for coral conservation. In: J. Thomas and P. Barnes (Eds.), Symposium on the Effects of Fishing Activities on Benthic Habitats: Linking Geology, Biology, Socioeconomics and Management. American Fisheries Society, Bethesda, Maryland, USA, pp. 819-832.
Grehan, A., Long, R. , Deegan, B. and O'Cinneide, M. , 2003. The Irish Coral Task Force and Atlantic Coral Ecosystem Study Report on Two DeepWater Coral Conservation Stakeholder Workshops. Marine Environment and Health Series, 11, 117.
Grigg, R.W., 2002. Precious corals in Hawaii: discovery of a new bed and revised management measures for existing beds. Marine Fisheries Review, 64(1), 13-20.
Gudmundsson, E. and Wessells, C.R., 2000. Ecolabeling seafood for sustainable production: implications for fisheries management. Marine Resource Economics, 15(2), 97-113.
Haigh, N., 1993, The precautionary principle and British environmental policy. Institute for European Environmental Policy, London.
Hall-Spencer, J., Allain, V. and Fossa, J. H., 2002. Trawling damage to Northeast Atlantic ancient coral reefs. Proceedings of the Royal Society of London Series B-Biological Sciences, 269(1490), 507-511.
Hannesson, R. , 2004. The Privatization of the Oceans. The MIT Press, USA.
Hatcher, B. G. and Schiebling, R. E. , 2001, What determines whether deep-water corals build reefs: do shallow reef models apply? In: J.H.M. Willison, J. Hall, S. Gass, E. Kenchington, M. Butler et al. (Eds.), First International Symposium on Deep-Sea Corals. Ecology Action Centre, Halifax, Nova Scotia, pp. 6 - 18.
Heifitz, J., 2002. Coral in Alaska: distribution, abundance, and species associations. Hydrobiologia, 471, 19-28.
Holland, D. and Schnier, K.E., 2006. Individual habitat quotas for fisheries. Journal of Environmental Economics and Management, 51(1), 72-92.
Holling, C.S., 1973. Resilience and stability of ecological systems. Annual Review of Ecological Systems, 4, 1-24.
Holling, C.S., Schindler, D.W., Walker, B.W. and Roughgarden, J., 1995, Biodiversity in the functioning of ecosystems: an ecological primer and synthesis. In: C. Perrings, K-G. Maler, C. Folke, B.O. Jansson and C.S. Holling (Eds.), Biodiversity loss: ecological and economic issues. Cambridge University Press, Cambridge.
Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P. et al., 2005. Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. Ecological Monographs, 75(1), 3-35.
Hughes, A.R. and Stachowicz, J.J., 2004, Genetic diversity enhances the resistance of a seagrass ecosystem to disturbance, Proceedings of the National Academy of Sciences, pp. 8998-9002.
Hughes, T.P., Bellwood, D.R., Folke, C., Steneck, R.S. and Wilson, J., 2005. New paradigms for supporting the resilience of marine ecosystems. Trends in Ecology and Evolution, 20(7), 380-386.
Husebø, A., Nottestad, L., Fossa, JH, Furevik, DM and Jorgensen, SB, 2002. Distribution and abundance of fish in deep sea coral habitats. Hydrobiologia, 471, 91-99.
Jensen, F. and Vestergaard, N., 2003. Prices versus quantities in fisheries models. Land Economics, 79(3), 415-425.
Johnston, R.J., Wessells, C.R., Donath, H. and Asche, F., 2001. Measuring consumer preferences for ecolabeled seafood: an international comparison. Journal of Agricultural and Resource Economics, 26(1), 20-39.

Jones, J.B., 1992. Environmental impact of trawling on the seabed: a review. New Zealand Journal of Marine and Freshwater Research, 26, 59-67.
Kenyon, W. and Nevin, C. , 2001. The use of economic and participatory approaches to assess forest development: a case study in the Etterick Valley. Forest Policy and Economics, 3, 69-80.
Kniivilä, M., 2006. Users and non-users of conservation areas: Are there differences in WTP, motives and the validity of responses in CVM surveys? Ecological Economics, 59(4), 530-539.
Knowler, D, 2002. A Review of Selected Bioeconomic Models with Environmental Influences in Fisheries. Journal of Bioeconomics, 4(2), 163-181.
Koenig, C.C., 2001. *Oculina* banks: habitat, fish populations, restoration, and enforcement, Florida State University.
Koslow, J.A., Gowlett-Holmes, K. , Lowry, J.K., O'Hara, T., Poore, G.C.B. et al., 2001. Seamount benthic microfauna off southern Tasmania: community structure and impacts of trawling. Marine Ecology Progress Series, 213, 111-125.
Koslow, T., 2003. Deep water fisheries and seamount conservation in Tasmanian Sea and Australia. Marine Environmental and Health Series No. 11, Marine Institute, Ireland, Galway.
Krieger, K.J., 2001, Coral *(Primnoa)* impacted by fishing gear in the Gulf of Alaska. In: J.H.M. Willison, J. Hall, S. Gass, E. Kenchington, M. Butler et al. (Eds.), First International Sympoium on Deep-Sea Corals. Ecology Action Centre, Halifax, Nova Scotia, pp. 106-116.
Krutilla, J.V., 1967. Conservation reconsidered. American Economic Review, 57, 778-786.
Lauck, T., Clark, C. W. , Mangel, M. and Munro, G, 1998. Implementing the precautionary principle in fisheries management through marine reserves. Ecological Applications, 8(1), 72-78.
Ledoux, L., Turner, R.K., Mathieu, L. and Crooks, S., 2001, Valuing ocean and coastal resources: practical examples and issues for further action, Global Conference on Oceans and Coasts at Rio+10: Assessing Progress, Addressing Continuing and New Challenges. UNESCO, Paris.
Long, R., 2007, Marine Resource Law. Thomson Roundhall, Dublin.
Long, R. and Grehan, A., 2002. Marine habitat protection in sea areas under the jurisdiction of a coastal member state of the European Union: the case of deep water coral conservation in Ireland. The International Journal of Marine and Coastal Law 17(2), 235-259.
Loomis, J., Kent, P., Strange, L., Fausch, K. and Covich, A., 2000. Measuring the total economic value of restoring ecosystem services in an impaired river basin: the results from a contingent valuation survey. Ecological Economics, 33, 103-117.
Lumsden, S.E., Hourigan, T.F., Bruckner, A.W. and Dorr, G. (Eds.), 2007. The state of deep coral ecosystems of the United States. NOAA Technical Memorandum CRCP-3. Silver Spring MD.
Lutringer, A. , Blamart, D., Frank, N. and Laberyrie, L., 2005, Paleotemperatures from deep-sea corals: scale effects. In: A Freiwald and J. Roberts (Eds.), Cold-water corals and ecosystems. Springer-Verlag, Berlin Heidelberg, pp. 1097-1108.
Lynne, G.D., Conroy, P. and Prochaska, F.J., 1981. Economic value of marsh areas for marine production processes. Journal of Environmental Economics and Management, 8, 175-186.
MacIsaac, K., Bourbannais, C., Kenchington, E., Gordon, D. and Gass, S. , 2001, Observations on the occurrence and habitat preference of corals in Atlantic Canada. In: J.H.M. Willison, J. Hall, S. E. Gass, E.L.R. Kenchington, M. Butler et al. (Eds.), First International Symposium on Deep-Sea Corals. Ecology Action Centre, Halifax, Nova Scotia, pp. 58 -75.
McAllister, D.E. and Alfonso, N., 2001, The distribution and conservation of deep-water corals on Canada's west coast. In: J.H.M. Willison, J. Hall, S. Gass, E. Kenchington, M. Butler et al. (Eds.), First International Symposium on Deep-sea Corals. Ecology Action Centre, Halifax, Nova Scotia, pp. 126 - 144.
Miller, C.A., 2001, Marine protected area framework for deep-sea coral conservation In: J.H.M. Willison, J. Hall, S. Gass, E. Kenchington, M. Butler et al. (Eds.), First International Symposium on Deep-Sea Corals. Ecology Action Centre, Halifax, Nova Scotia, pp. 145-155.
Moberg, F. and Folke, C. , 1999. Ecological goods and services of coral reef ecosystems. Ecological Economics, 29(2), 215-233.
Mortensen, P.B., Buhl-Mortensen, L., Gordon, D.C., Fader, G.B., McKeown, D.L. et al., 2005, Effects of fisheries on deepwater Gorgonian corals in the Northeast Channel, Nova Scotia. In: P.W. Barnes and J.P. Thomas (Eds.), Benthic habitats and the effects of fishing. American Fisheries Society, Bethesda, Maryland.
Mullineaux, L.S. and Mills, S.W., 2005, Coral Gardens in the Dark Depths, Oceanus, pp. 22 - 27.
Myers, N., 1992. Population/environment linkages: discontinuities ahead. Ambio, 21, 116-118.
Oelofsen, E., 1999. Fisheries management: the Namibian approach. ICES Journal of Marine Science, 56, 999-1004.

Pauly, D. , Christensen, V., Dalsgaard, J., Froese, R. and Torres, F. Jr., 1998. Fishing down marine food webs. Science, 279, 861-863.

Pearce, D.W. and Turner, R.K., 1990, Economics of natural resources and the environment. Harvester Wheatsheaf, Hemel Hampstead, UK.

Perman, R., Ma, Y., McGilvray, J. and Common, M., 2003, Natural resource and environmental economics. Pearson Education, New York, Longman.

Perrings, C. and Pearce, D.W., 1994. Threshold effects and incentives for the conservation of biodiversity. Environmental and Resource Economics, 4, 13-28.

Peterson, C.H., Summerson, H.C., Thomson, E., Lenihan, H.S., Grabowski, J. et al., 2000. Synthesis of linkages between benthic and fish communities as a key to protecting essential fish habitat. Bulletin of Marine Science, 66(3), 759-774.

Puglise, K.A., Brock, R.J. and McDonough, J.J. , 2005, Identifying critical information needs and developing institutional partnerships to further the understanding of Atlantic deep-sea coral ecosystems. In: A Freiwald and J.M. Roberts (Eds.), Cold-water corals and ecosystems. Springer-Verlag, Berlin-Heidelberg, pp. 1129 - 1140.

Reed, J.K., 2002. Deep-water Oculina coral reefs of Florida: biology, impacts and management. Hydrobiologia, 471, 43-55.

Reusch, T.B.H., Ehlers, A., Hammerli, A. and Worm, B., 2005, Ecosystem recovery after climatic extremes enhanced by genotypic diversity, Proceedings National Academy Science, USA, pp. 2826-2831.

Risk, M.J., Hall-Spencer, J. and Williams, B., 2005, Climate records from the Faroe-Shetland Channel using *Lophelia pertusa* (Linnaeus, 1758). In: A. Freiwald and J.M. Roberts (Eds.), Cold-Water Corals and Ecosystems. Springer Verlag, Berlin Heidelberg, pp. 1097-1108.

Roberts, J. Murray, Wheeler, Andrew J. and Freiwald, Andre, 2006. Reefs of the Deep: The Biology and Geology of Cold-Water Coral Ecosystems. Science, 312(5773), 543-547.

Rogers, A.D., 1999. The biology of Lophelia pertusa (Linnaeus, 1758) and other deep-water reef-forming corals and impacts from human activities. International Review of Hydrobiology, 84(4), 315 - 406.

Rosenberg, A., Bigford, T.E., Leathery, S., Hill, R.L., Bickers, K. et al., 2000. Ecosystem approaches to fishery management through essential fish habitat. Bulletin of Marine Science, 66(3), 535-542.

Sanchirico, J.N., 2008. An empirical approach to ecosystem-based fishery management. Ecological Economics, 64(3), 586-596.

Sathirathai, S. and Barbier, E. B., 2001. Valuing Mangrove Conservation in Southern Thailand. Contemporary Economic Policy, 19(2), 109-122.

Shester, G. and Ayers, J., 2005, A cost effective approach to protecting deep-sea coral and sponge ecosystems with an application to Alaska's Aleutian Islands region. In: A Freiwald and J.M. Roberts (Eds.), Cold-water corals and ecosystems. Springer-Verlag, Berlin Heidelberg, pp. 1151-1169.

Simpson, R.D., Sedjo, R.A. and Reid, J.W., 1996. Valuing biodiversity for use in pharmaceutical research. Journal of Political Economy, 104(1), 163-185.

Soderqvist, T., Eggert, H., Olsson, B. and Soutukorva, A., 2005. Economic valuation for sustainable development in the Swedish coastal zone. Ambio, 34(2), 169-175.

Solow, A., Polasky, S. and Broadus, J., 1993. On the measurement of biological diversity. Jounal of Environmental Economics and Management, 24, 60-68.

Spaninks, F. and van Beukering, P., 1997. Economic valuation of mangrove ecosystems: potential and limitations. CREED working paper, no. 14.

Spash, C. and Hanley, N., 1995. Preference, information and biodiversity preservation. Ecological Economics, 12(3), 191-208.

Steneck, R.S., Graham, M.H., Bourque, B.J., Corbett, D., Erlandson, J.M. et al., 2002. Kelp forest ecosystem: biodiversity, stability, resilience and their future. Environmental Conservation, 29(4), 436-459.

Steneck, R.S., Vavrinec, J. and Leland, A.V., 2004. Accelerating trophic level dysfunction in kelp forest ecosystems of the western North Atlantic. Ecosystems, 7(4), 323-331.

Sumaila, Ussif Rashid, Guenette, Sylvie, Alder, Jackie and Chuenpagdee, Ratana, 2000. Addressing ecosystem effects of fishing using marine protected areas. ICES J. Mar. Sci., 57(3), 752-760.

Taylor, P., 1991. The precautionary principle and the prevention of pollution ECOS, 12(4), 41-46.

Tilman, D., Reich, P.B. and Knops, J.M.H., 2006. Biodiversity and ecosystem stability in a decade-long grassland experiment. Nature, 441(7093), 629-632.

Turley, C.M., Roberts, J.M. and Guinotte, J.M., 2007. Corals in deep water: will the unseen hand of ocean acidification destroy cold-water coral ecosystems? Coral Reefs, 26, 445-448.
Turpie, J. K., 2003. The existence value of biodiversity in South Africa: how interest, experience, knowledge, income and perceived level of threat influence local willingness to pay. Ecological Economics, 46, 199-216.
Vane-Wright, R., Humphries, C.J. and Williams, P.H., 1991. What to protect? Systematics and the agony of choice. Biological Conservation, 55(3), 235-254.
Walsh, R.G., Loomis, J.B. and Gillman, R.A., 1984. Valuing option, existence, and bequest demands for wilderness. Land Economics, 60(1), 14-29.
Watling, L. and Norse, E.A., 1998. Disturbance of the seabed by mobile fishing gear: a comparison to forest clearcutting. Conservation Biology, 12(6), 1180-1197.
Watling, L. and Langton, R., 1994. Fishing, habitat disruption and biodiversity loss. EOS Transaction and American Geophysical Union, 75(3).
Weisbrod, B., 1964. Collective consumption services of individual consumption goods. Quarterly Journal of Economics, 78(3), 471-477.
Weitzman, Martin L., 1998. The Noah's Ark Problem. Econometrica, 66(6), 1279-1298.
Wessells, C.R., Johnston, R.J. and Donath, H., 1999. Assessing consumer preferences for ecolabeled seafood: the influence of species, certifier and household attributes. American Journal of Agricultural Economics, 81(5), 1084-1089.
Whitehead, J.C., Blomquist, G.C., Hoban, T.J. and Clifford, W.B., 1995. Assessing the validity and reliability of contingent values: a comparison of on-site users, off-site users, and non-users. Journal of Environmental Economics and Management, 29, 238-251.
Wilson, J.B. , 2001, Lophelia 1700 to 2000 and beyond. In: J.H.M. Willison, J. Hall, S. Gass, E. Kenchington, M. Butler et al. (Eds.), First International Symposium on Deep-Sea Corals. Ecology Action Centre, Halifax, Nova Scotia, pp. 1 - 5.
Witherell, D. and Coon, C. , 2001, Protecting gorgorian corals off Alaska from fishing impacts. In: J.H.M. Willison, J. Hall, S. Gass, E. Kenchington, M. Butler et al. (Eds.), First International Symposium on Deep-sea Corals. Ecology Action Centre, Halifax, Nova Scotia, pp. 117 - 125.
Young, M.D., 1995. The design of fishing-right systems - The New South Wales experience. Ocean & Coastal Management, 28(1-3), 45-61.
Young, M.D., 1999. The design of fishing-right systems - The New South Wales experience. Ecological Economics, 31(2), 305-316.