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## Review

# Preserving deep-sea natural heritage: Emerging issues in offshore conservation and management

Andrew J. Davies<sup>a,\*</sup>, J. Murray Roberts<sup>a</sup>, Jason Hall-Spencer<sup>b</sup>

<sup>a</sup>Scottish Association for Marine Science, Dunstaffnage Marine Laboratory, Oban, Argyll PA37 1QA, UK

<sup>b</sup>Marine Biology and Ecology Research Centre, School of Biological Sciences, University of Plymouth, Drake Circus, Plymouth PL4 8AA, UK

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## ABSTRACT

Human activity in the deep sea is extending ever deeper, with recent research showing that this environment is more sensitive to human and natural impacts than previously thought. Some deep-water fish stocks have collapsed and fishing methods such as bottom trawling have raised international concern over the habitat damage they cause. It is likely that in its current form, deep-sea fishing is unsustainable. Diminishing reserves of hydrocarbons in shallow water are pushing exploration and production into deeper waters, which may cause damage to little known deep-sea habitats. The deep sea is also proposed as an environment where anthropogenic carbon dioxide could be stored to minimise the effect of its release into the atmosphere. At the same time, rising atmospheric carbon dioxide levels may be altering the chemical equilibrium of the global ocean by lowering pH. Many countries are now beginning to designate some deep-sea habitats as marine protected areas in measures to reduce the damage caused by fishing and other anthropogenic activities. This review examines these current and emerging issues in deep-sea conservation and discusses conservation status and the designation of protected areas. The enforcement of protected areas using satellite tracking of vessels is discussed and applied to an internationally agreed deep-water conservation area, which aims to protect cold-water coral habitats on the Darwin Mounds in the north east Atlantic Ocean.

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\* Corresponding author. Tel.: +44 0 1631 559019; fax: +44 0 1631 559001.

E-mail address: [andrew.davies@sams.ac.uk](mailto:andrew.davies@sams.ac.uk) (A.J. Davies).

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## 1. Introduction

Waters of over 1000 m deep cover an estimated 62% of the surface of the earth, forming one of the last great wilderness areas on the planet (Roberts, 2002a). Only within the last few decades has technology advanced sufficiently to map the deep sea effectively (Diaz et al., 2004), revealing extensive and diverse habitats of conservation interest and economic value (Tunnicliffe, 1991; Rogers, 1999; Hovland et al., 2002; Roberts et al., 2005b). Concurrently, human activity, most notably fishing, has continually expanded from coastal waters with great increases in the amount of activity and depths of operations since the 1950s (Morato et al., 2006). Fishing activities are increasingly affecting the deep ocean, to the extent that quantifiable changes in fish stocks (Baum et al., 2003; Morato et al., 2006) and damage to benthic habitats (Roberts et al., 2000; Koslow et al., 2001; Hall-Spencer et al., 2002; Gage et al., 2005; Wheeler et al., 2005) have now been recorded around the world.

Many deep-sea areas (>200 m after Gage and Tyler, 1991) are found beyond the exclusive economic zone of individual nations on the High Seas. Under the Third United Nations Convention on the Law of the Sea (UNCLOS; UN, 1982), the High Seas are an open-access commons where all nations have the right to engage in fishing (UNCLOS, Article 87). With no clear ownership or jurisdiction, these areas are potentially vulnerable to over-exploitation and unregulated activities. However, there are provisions for conservation within the UNCLOS, with nations required to cooperate in the conservation of living marine resources of the High Seas (UNCLOS, Articles 116–120). One development from the UNCLOS (and subsequent supplementary agreement on the implementation of Part XI in 1996; UN, 1996) was the genesis of the International Seabed Authority which regulates all solid, liquid or gaseous mineral resources at or beneath the seabed in the High Seas. Unfortunately, the majority of legal provisions laid out within the UNCLOS have not yet been developed into a sufficiently comprehensive framework to efficiently manage the majority of resources in the High Seas (FAO, 2004).

The lack of incentives for nations to constrain fishing effort and comply with conservation measures on the High Seas raises concern over the absence of effective international measures for assessing and protecting newly discovered habitats (FAO, 2004). The deep sea contains many distinct habitats including hydrothermal vents, cold seeps, seamounts and cold-water coral reefs amongst the expanses of abyssal muds, themselves home to a notably rich animal community (Grassle and Maciolek, 1992; Gage, 1996).

## 2. Current threats

Anthropogenic threats can be either direct, such as fishing or hydrocarbon exploitation or indirect, such as increasing

atmospheric CO<sub>2</sub> levels leading to changes in the chemistry of the ocean. This review summarises emerging threats to deep-sea ecosystems (e.g. climate change) and provides recent evidence to update the threats reviewed by Glover and Smith (2003) (e.g. fisheries, oil and gas and carbon capture and storage). Several significant threats are omitted due to coverage in earlier reviews (e.g. dumping and mining) but this does not mean these threats are not severe. The mining of deep-sea mineral resources is imminent, with licenses already being provided by the International Seabed Authority (ISA, 2006). This review closes by discussing protection and enforcement initiatives using a cold-water coral ecosystem case study.

### 2.1. Climate change and ocean acidification

Numerous studies have reported changes in global climate (e.g. IPCC, 2001) with many significant effects recorded in organisms throughout the world (e.g. Scheffer et al., 2001; Walther et al., 2002). Previously, the deep sea was thought to be buffered against the effects of surface-driven cycles and impacts (Menzies, 1965), but modern research has now shown the deep-sea to be susceptible to changes at the surface. These observations include variation in physical processes such as currents and seasonal influxes of particulate matter, as well as variation in ecological processes such as reproduction, growth rate and oxygen consumption (e.g. Tyler, 1988; Gooday, 2002). Several impacts on the surface could have implications for deep-sea habitats and organisms. Changes in surface productivity and environmental variables such as temperature or salinity could potentially influence deep-sea species distribution, abundance and behaviour (Levin et al., 2001; Ruhl and Smith, 2004). Increasing fresh water input from terrestrial sources may disrupt thermohaline circulation, changing ocean circulation patterns and affecting water temperature and density (Hansen et al., 2004). In the deep Eastern Mediterranean Sea, between 1992 and 1994, abrupt changes in water temperature and density led to a rapid change in the nematode community. There was a significant decrease in the abundance of some genera but an overall increase in species diversity (Danovaro et al., 2004). Even abrupt, short-term changes in environmental conditions at the seafloor may have long-term effects on the composition and functioning of deep-sea ecosystems.

Surface productivity, especially from phytoplankton and zooplankton has been strongly linked to climatic fluctuations (Edwards et al., 2001; Hawkins et al., 2003; Schippers et al., 2004; Pitois and Fox, 2006). In the North East Atlantic phytodetrital deposits during spring and early summer deliver an estimated 2–4% of spring-bloom surface production to the seafloor (Gooday, 2002). This usually consists of fresh phytoplankton and a high proportion of diatoms as well as other

biogenic particles, forming an important and recurrent source of matter to the deep sea (Lampitt, 1985; Turley et al., 1995). Within days of the arrival of phytodetritus, there are large increases in the biomass and activity of protozoans and bacteria (Graf, 1989; Soltwedel, 1997). Over the following weeks, increases in the abundance and biomass of Foraminifera, metazoan meiofauna and epibenthic megafauna have been recorded (Pfannkuche, 1992, 1993; Bett et al., 2001; Hughes and Gage, 2004). Changes in the distribution of productivity on the surface would likely alter the distribution of species on the seafloor. In the deeper waters of the North Pacific, declines in food supply from surface waters has led to decreased community oxygen consumption and continued disruption is expected to lead to changes in the structure and dynamics of these communities (Smith and Kaufmann, 1999). One enigmatic group of deep-sea organisms, cold-water corals, have been strongly linked with areas of high surface productivity and accelerated currents (Rogers, 1999; Guinotte et al., 2006; Roberts et al., 2006b), any disruption to surface patterns may have negative effects on these and other deep-sea habitats (Ruhl and Smith, 2004).

A key emerging issue has been the recent prediction that the chemistry of the oceans will change due to rising concentrations of anthropogenic CO<sub>2</sub> in the atmosphere. The global ocean is a substantial reservoir of carbon, containing about 38,000 gigatonnes of carbon (Gt C), compared to 700 Gt C in the atmosphere and 2000 Gt C in terrestrial systems (Brovkin et al., 2002). When atmospheric carbon in the form of CO<sub>2</sub> is absorbed by the ocean it becomes reactive and produces hydrogen ions due to an initial reaction with H<sub>2</sub>O, forming carbonic acid (H<sub>2</sub>CO<sub>3</sub>), this has the effect of making the ocean more acidic (Raven et al., 2005). Globally, the surface ocean has an average pH of 8.2 units (±0.3 units due to local, regional and seasonal factors; Raven et al., 2005). Observations and models have found that the average pH of the ocean has declined by 0.1 units since pre-industrial times and is projected to decrease by 0.3–0.7 units under the IPCC IS92a scenario by the end of the century (Brewer, 1997; Caldeira and Wickett, 2003). Such large scale changes are greater than any over the past 300 million years (Caldeira and Wickett, 2003). The age of “ocean acidification” may have begun.

Current geochemical models predict that as well as acidification, further changes will occur in chemistry of the oceans (Caldeira and Wickett, 2005; Orr et al., 2005). Carbon that is dissolved in seawater exists in three main inorganic forms, collectively known as dissolved inorganic carbon (DIC). (1) Aqueous CO<sub>2</sub> (CO<sub>2</sub>(aq)), (2) bicarbonate (HCO<sub>3</sub><sup>-</sup>) and (3) carbonate ions (CO<sub>3</sub><sup>2-</sup>). The most abundant form is bicarbonate, followed by carbonate ions and then CO<sub>2</sub>(aq). These forms of dissolved carbon act as a natural buffer and maintain the pH of seawater within natural limits (see Annex 1 in Raven et al., 2005). An acidified ocean is characterised by an increase in CO<sub>2</sub>(aq), hydrogen ions and HCO<sub>3</sub><sup>-</sup> and a decrease in pH and CO<sub>3</sub><sup>2-</sup> concentration. The effect of ocean acidification is most pronounced at the surface, with organisms in shallow waters likely to be affected first (see reviews by Raven et al., 2005; Hauge et al., 2006; Kleypas et al., 2006). Initially, the photosynthetic rate of some phytoplankton species may increase due to increased CO<sub>2</sub> availability (Beardall and Raven, 2004; Schip-

pers et al., 2004) but this could be offset in the longer term with impacts on growth and community composition of phytoplankton (Raven et al., 2005). However, there is a general lack of conclusive data for the majority of phytoplankton groups.

Coccolithophores and other calcifying planktonic organisms are an important source of calcium carbonate (CaCO<sub>3</sub>) to the deep-sea floor and are important in marine carbon cycles and ocean-atmosphere CO<sub>2</sub> exchange (Milliman, 1993). Calcifying marine plankton grown under simulated future acidified ocean conditions are unlikely to maintain present-day production levels, due to slowing of calcification and growth (Riebesell et al., 2000; Zondervan et al., 2001). Surface calcification is a major source of ballast for the transport of organic carbon to the deep-sea. Under modelled ocean conditions the production of these calcifying organisms could decrease by as much as 50% from pre-industrial values 250 years from now (Heinze, 2004). Not only would this remove a major carbon sink and an important source of organic carbon to the deep-sea floor but it would impact the ability of the ocean to buffer CO<sub>2</sub> absorption. This may lead to a potential increase in carbon uptake in surface waters under IS92a scenarios (Riebesell et al., 2000).

A major group of deep-sea organisms that could be affected by ocean acidification are cold-water corals (Turley et al., in press). Cold-water coral reefs are important habitat for a wide variety of species (Roberts et al., 2006b), and may form essential fish nursery and feeding grounds (Husebø et al., 2002). The present day distribution of cold-water coral reefs largely reflects the parts of the ocean that are supersaturated with respect to aragonite (Guinotte et al., 2006). In seawater, CaCO<sub>3</sub> can be fixed into two forms, aragonite and calcite (calcite is the less soluble of the two), the stability of each form is governed by the amount of free CO<sub>3</sub><sup>2-</sup>, temperature, salinity, pressure and a stoichiometric solubility product for each form (Feely et al., 2004). The depth boundary where seawater becomes undersaturated with respect to aragonite and calcite is known as the saturation horizon. Organisms that utilise CaCO<sub>3</sub> skeletons or plates become hindered in growth and biogenic structures become vulnerable to dissolution below the saturation horizon (Feely et al., 2004; Orr et al., 2005; Guinotte et al., 2006; Kleypas et al., 2006). The majority of records of reef framework-forming cold-water corals are from the North Atlantic Ocean, where the aragonite saturation horizon (ASH) is much deeper than in the Pacific (Orr et al., 2005). The shoaling of the ASH in the Pacific restricts the formation of aragonite skeletons to shallower waters. In the Pacific, coral communities are dominated by solitary octocorals and hydrocorals, a contrast to the large scleractinian reef structures in the North Atlantic (Guinotte et al., 2006). Models of the projected aragonite saturation show that ca. 70% of current known cold-water coral locations could become undersaturated by 2099, with the vast majority of the remaining 30% restricted to the North Atlantic (Guinotte et al., 2006).

Currently, only about 7% of the total anthropogenic CO<sub>2</sub> found in the ocean has penetrated deeper than 1500 m (Sabine et al., 2004), but the effect of climate change in the deep sea has now been recorded in several studies. The deep-sea areas or species that are likely to be most affected by climate change are those that rely on food supply from surface waters, calcifying marine organisms or those that have

specific environmental tolerances to variables such as salinity or temperature (Smith and Kaufmann, 1999; Danovaro et al., 2001, 2004; Ruhl and Smith, 2004; Guinotte et al., 2006). Yet, there is still a clear need for comprehensive research into the effect of climate change in the deep sea.

## 2.2. Fisheries

In the 1950s and 1960s the deep-water fishing industry expanded rapidly following a decline in catches from the traditional fisheries of shelf seas (Koslow et al., 2000; Roberts, 2002a). This shift in effort also coincided with improvements to fishing gear and the development of large, refrigerated vessels capable of spending extended periods at sea. Since 1950 the mean depth of fishing in the North Atlantic Ocean has increased by 32 m per decade as fishermen continually seek to fish unexploited areas which usually lie in deeper waters (Morato et al., 2006). This coincided with recorded declines in the diversity of some fisheries. For example, the diversity of High Seas predatory fishes are estimated to have declined between 10 and 50% since 1950 (Worm et al., 2005).

Surprisingly, catch statistics from deep-sea fisheries have appeared relatively stable until recently, disguising the decline of individual species as fishermen serially exploit unfished species when faced with diminishing catches (Koslow et al., 2000). It is now evident that many deep-sea fisheries have collapsed or are beginning to show warning signs of population decline (Baum et al., 2003; Devine et al., 2006). The inherently low recovery potential of deep-sea fish and the practice of serial depletion (e.g. targeting spawning aggregations of fish; Roberts, 2002a) have over-exploited some fisheries to a level which cannot be sustained (Berkes et al., 2006). Some deep-sea fish species now meet the World Conservation Union's (IUCN) criteria for being critically endangered (Devine et al., 2006).

Fishing methods in the deep sea are largely unselective of the species caught, leading to a high amount of discarded by-catch. Deep-sea tangle and gillnet fishing has been the recent focus of several European Union studies (Pawson, 2003; Hareide et al., 2005). Currently, approximately 50 vessels in the North East Atlantic target mostly monkfish and deepwater sharks (Hareide et al., 2005). Whilst this fishery is largely unregulated and poorly understood, there are concerns over the "ghost" fishing of lost nets and the high amount of by-catch and discards relative to the amount of catch (Hareide et al., 2005). In shallower waters (<100 m), the majority of nets rapidly lose their catching efficiency within a month of being lost (Pawson, 2003). Nets lost in deeper waters may continue to fish over much longer periods due to the relatively low current energy. For example, nets between 117 and 135 m continued to catch monkfish efficiently for 135 days, with the nets ceasing to capture fish after 224 days (Sancho et al., 2003). Currently, no substantial long-term research has been conducted on the effect of "ghost" fishing nets in the deep sea, but these nets are expected to stabilise to approximately 20% of the initial catch after 45 days (Humborstad et al., 2003) but may fish for periods of at least 2–3 years and may even do so for longer periods (Furevik and Fosseidengen, 2000).

In terms of global effort, reported deep-water landings accounted for 6.5% of total fisheries capture in 2002 (4.7%

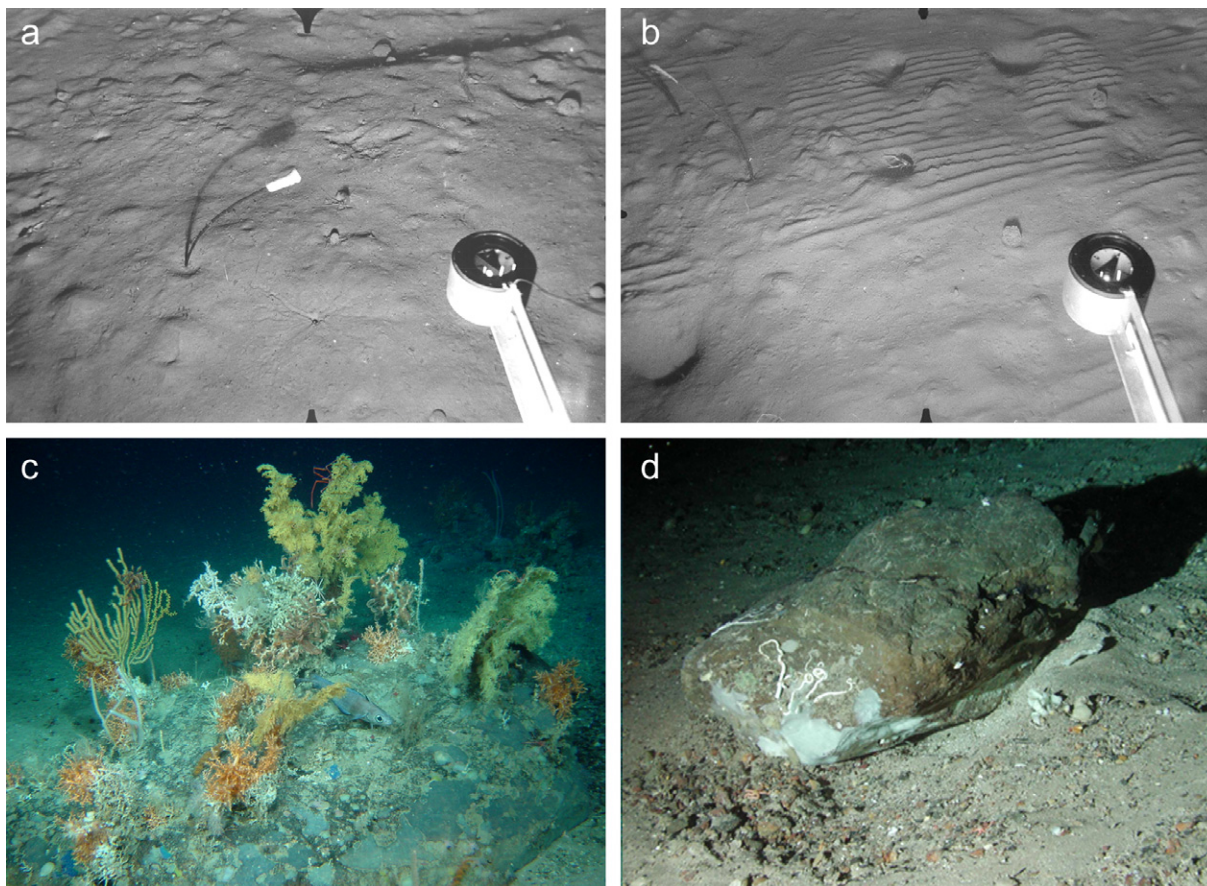
excluding China due to uncertainty in reporting statistics), a small proportion of a global industry worth between \$70 and \$80 billion (USD) year<sup>-1</sup> (FAO, 2004). Deep-sea bottom trawling, one of the most physically damaging fishing methods, is not likely to exceed a value of \$300–400 million annually and contributes an estimated 0.2–0.25% of global fisheries capture weight (Gianni, 2004). The trawls used in the deep sea are modified forms of shallow water trawls, sometimes fitted with rock-hopper gear and kept open by large otter boards (or trawl doors) that can weigh up to a tonne each. On some occasions the base of the net is reinforced by chains, which serve to strengthen the net, adding weight to maintain close contact with the seafloor. On an average fishing excursion, these trawls are deployed for approximately 4 h, sweeping 20–30 km of seabed at 3–5 knots. There can be up to 5 hauls day<sup>-1</sup>, and fishing excursions often last approximately 10 days, potentially covering up to 100 km<sup>2</sup> of seafloor (Hall-Spencer et al., 2002). The practice of bottom trawl fishing is causing international concern due to the damage that occurs to benthic habitats during the passage of a trawl (e.g. Hall-Spencer et al., 2002; Gage et al., 2005).

One of the key differences between deep sea and shallow waters is the persistence of trawling damage. The deep sea in general is characterised by low current speeds and sedimentation rates resulting in the long-term persistence of trawling damage to the seabed (Gage et al., 2005), although there are some exceptions (e.g. coarse sediments that are less stable than muddy, sandy sediments; Roberts et al., 2000; Ball et al., 2000). Regardless of the ecosystem, trawling scrapes and ploughs the sediment, creating areas of intense disturbance (Roberts et al., 2000, see Fig. 1) and in the process rolls large boulders, removing any epifauna (McAllister et al., 1999; Gage et al., 2005). The passage of the trawl can also resuspend large quantities of sediment (Palanques et al., 2001, 2006), which smothers organisms and reduces the abundance of some components of the benthos (Yamazaki et al., 1997). The most significant and potentially persistent effect of trawling may be the damage of non-target epibenthic species such as sponges, xenophyophores and cold-water corals (Pitcher et al., 2000; Roberts et al., 2000; Koslow et al., 2001; Fosså et al., 2002; Hall-Spencer et al., 2002; Wheeler et al., 2005), but the long-term effects on marine organisms other than corals are largely unknown (Gage et al., 2005).

## 2.3. Oil and gas

Oil and gas extraction has been undertaken for many years in shelf seas but in recent years production has started to diminish (Kemp and Stephen, 2005). With continually increasing consumer demand and extraction costs some international hydrocarbon companies are now under significant pressure to discover new reserves (Hallock et al., 2004). A promising source of new reserves may be in deeper waters. Exploitation of deep-sea hydrocarbons began in the Gulf of Mexico in 1979, with six producing wells by 1992. This number rose to 17 by 1997 and increased exponentially to 118 by March 2006 mostly between 500 and 1500 m water depths (French et al., 2006). The Gulf of Mexico now contains the deepest well to date, the Toledo well drilled at 3051 m water depth in late 2003. There is further potential for large reserves to be exploited in deep-





**Fig. 1** – Images showing the effects of bottom trawling on benthic habitats. (a) Untrawled soft sediment with intact glass sponge (*Hyalonema* sp.), 1295 m. (b) Trawled soft sediment with broken glass sponge stalk, 1298 m. See Roberts et al., 2000. (c) Untrawled dropstone with diverse coral and sponge epifauna, 970 m Porcupine Bank (Photo courtesy of IFEMER and AWI). (d) Trawled dropstone, 590 m. See Gage et al., 2005.

sea areas off the coasts of South Africa, West Africa, Brazil and the Indo-Pacific region including Malaysia, Indonesia, Brunei and India (Zhu, 2004). In many parts of the world, such as the North Atlantic, offshore activities are strongly regulated to reduce pollution and the amount of waste deposited on the seabed (Hartley, 2005). For example, in the UK, regulation has led to noticeable reductions in the levels of discharges recorded, including carbon dioxide (CO<sub>2</sub>) emissions, chemical discharges and contaminated drill cuttings (UKOOA, 2005).

The ocean currents in the deep-waters surrounding deep-sea exploration and production facilities would likely disperse the contaminants released. This would dilute the impact on deep-sea species and some species thought to be highly sensitive to drilling discharges may in fact show some resilience. For example, the cold-water coral *Lophelia pertusa* has been found growing on oil rigs in the North Sea (Bell and Smith, 1999; Roberts, 2002b; Gass and Roberts, 2006, see Fig. 2). The majority of colonies discovered on these rigs are in good condition. However, those observed growing close to discharge chutes have shown signs of contamination with some colonies suffering upwards of 30% polyp mortality (Gass and Roberts, 2006). It is probable that any discharges from deep-sea rigs at the surface or in the water column would disperse in the surrounding ocean, but discharges at the seabed may persist due to the low dispersal that characterises many deep-sea

habitats (UKOOA, 2002). If discharges do occur close to the seafloor, it is likely that smothering and potential mortality to epibenthic species such as corals would occur (Rogers, 1999). The lack of basic information on the biology and ecology of deep-sea species emphasises the need for research on the impacts of hydrocarbon exploitation in deeper waters.

#### 2.4. Carbon capture and storage

At present, global estimates suggest up to 25 billion tonnes of CO<sub>2</sub> are released into the atmosphere each year (Stephens and Van der Zwaan, 2005). One approach to mitigate rising CO<sub>2</sub> levels may be to capture and store CO<sub>2</sub> (referred to as carbon sequestration or carbon capture and storage – CCS) rather than releasing it into the atmosphere. This is an emerging technology that is currently expensive and difficult to implement (Stephens and Van der Zwaan, 2005). Carbon dioxide could be stored in underground geological formations including depleted oil and gas fields, deep aquifers, unminable coal seams and deep-sea sediments (Holloway, 2005; House et al., 2006). There is also the potential for liquefied CO<sub>2</sub> to be stored in the deep sea, achieved by injecting liquefied CO<sub>2</sub> into great depths. In November 2006, Contracting Parties to the London Convention (1972 and 1996) adopted amendments to Annex 1 of the 1996 Protocol to the Convention on the Prevention of



**Fig. 2 – Dome-shaped colonies of *Lophelia pertusa* (red and white colonies), observed growing on an oil platform in the North Sea (Photo courtesy of Lundin Petroleum Ltd.). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)**

Marine Pollution by Dumping of Wastes and Other Matter, 1972 (London Convention). This forms the legal basis allowing the sequestration of CO<sub>2</sub> beneath the seabed.

It is likely that high concentrations of CO<sub>2</sub> at injection points would have the most profound effects on deep-sea organisms (Barry et al., 2004). As currents disperse concentrated CO<sub>2</sub>, the lethal effects would most likely diminish with increasing distance from the source (Thistle et al., 2005). *In situ* experiments show the effects on deep-sea dwelling organisms can be profound (Seibel and Walsh, 2003). Benthic infauna and epifauna suffer respiratory stress and direct mortality from acidosis (Barry et al., 2004; Thistle et al., 2005, 2006). Mobile species may be able to move away from the CO<sub>2</sub> source and thus only suffer temporary effects such as loss of consciousness (Brewer et al., 1999; Tamburri et al., 2000). Longer-term exposure may be lethal even for mobile species, but the observed effects do vary widely between species (Ishimatsu et al., 2005). The benefits gained by storing carbon dioxide in this way needs to be assessed in relation to the hazards it poses to deep-sea life (Barry et al., 2004).

### 2.5. Summary of impacts

With so much evidence of anthropogenic impacts on the surface, it was only a matter of time before researchers noted a response in the deep sea (e.g. Thiel, 2003; Tyler, 2003; Glover and Smith, 2003). Destructive fishing practices, the serial removal of fish, the hunt for hydrocarbons and climate change are affecting deep-sea communities now. The lack of quantitative information from the deep-sea affects the conclusions that can be drawn on the extent, severity and persistence of these impacts. Therefore research now needs to shift from

locating and describing disturbance to quantifying the impacts over temporal and spatial scales. This information is especially important for the development of truly effective conservation measures. Whether this is a realistic proposition in the deep sea is a question that remains to be answered. However, some of these impacts could be mitigated or even avoided by the development of appropriate regulatory controls, the establishment of relevant protected areas and improved awareness of the deep-sea environment among both policy makers and the general public.

### 3. Protecting deep-sea biodiversity and ecosystems

Marine protected areas have been established to protect biodiversity and enhance fisheries in coastal areas throughout the world (Gell and Roberts, 2003; Halpern, 2003; Roberts et al., 2005a). These areas provide refuges for populations of exploited and vulnerable species or habitats by exerting strong control over human activity (Gell and Roberts, 2003). Such as placing restrictions on fishing gear, oil and gas prospecting or other human activities such as dumping. Many coastal marine protected areas have shown significant benefits since their establishment. Fisheries often increase adjacent to the reserves due to the emigration of adults and juveniles and the export of pelagic eggs and larvae (Bohnsack, 1998; Murawski et al., 2000; Tawake et al., 2001). Inside reserves, populations of many species often increase in abundance and individuals live longer with increased reproductive potential (Bohnsack, 1990; Gell and Roberts, 2003; Halpern, 2003). Sessile species that are vulnerable to physical damage or harvesting can recover. For example, on



the Georges Bank in the Gulf of Maine, populations of echinoderms, hydroids and seafans have increased in areas closed to trawling (NRC, 2002). It is likely that deep-sea protected areas would respond in a similar way although ecosystem connectivity and recovery timescales are less well understood. The process of applying protected area status to the deep sea represents a different set of challenges to those faced in terrestrial and coastal ecosystems.

Existing marine reserves have several key limitations, the most significant of which are size and location (Boersma and Parrish, 1999). The degree of protection a reserve can offer to a species can be assessed by its dispersal capability and how restricted the species is to a particular site (Kenchington, 1990). Species with low dispersal that are restricted to small areas are likely to be well protected by small reserves. However, highly mobile species may require extremely large reserves to provide adequate protection. Some estimates suggest that 50–90% of the total utilised habitat is needed (Clark, 1996; Lauck et al., 1998). The initial theory behind designing marine protected areas was developed for terrestrial systems and adapted for the coastal realm (Soulé and Terborgh, 1999; Carr et al., 2003). The designers of a reserve faces three immediate questions: (1) How large must the reserve be? (2) How many must there be? And (3) Where should they be located? (Carr et al., 2003). There are no single answers shared between terrestrial or coastal environments (Carr et al., 2003) and the answers are likely to be different again for deep-sea habitats and species. Especially since, significant legal, scientific and socio-economic issues face the development of effective deep-sea marine protected areas.

Although many deep-sea protected areas are likely to be within the exclusive economic zones of individual countries, there are many vulnerable species and habitats offshore on the High Seas. In these multiple user and multiple jurisdictional areas, the lack of effective legal frameworks makes the implementation and enforcement of protected areas more difficult (Gubbay, 1998). For example, it is currently possible for fishing vessels to avoid established regional fisheries management regulations by operating under flags of convenience (Gianni and Simpson, 2005). The countries to which vessels are registered are responsible for the actions of these vessels on the High Seas, but often exercise little or no control (Gianni and Simpson, 2005). These countries are often not registered with regional fisheries management organisations and currently, they cannot be held legally responsible for fishing activity under any regional management regulations. One example is in the Northeast Atlantic where only fishing vessels from Contracting Parties with the North East Atlantic Fisheries Commission (NEAFC) are legally bound to operate within established legalisation, such as effort reduction initiatives, no take zones and closed areas. Vessels flying flags of convenience from countries that are not registered with NEAFC and violate these regulations are not prosecutable under current legalisation. This form of unregulated fishing limits the ability of managers to enforce closed areas in international waters.

Scientific knowledge of the deep sea is continually improving but significant gaps remain. In particular, little is known about the dispersal of many deep-sea organisms, which has

clear consequences for the placement and size of effective reserves (Shanks et al., 2003). Mapping coverage in the deep sea is low, with the majority of effort targeted towards areas containing clearly identifiable habitats such as cold-water coral reefs (e.g. Roberts et al., 2005b). Current distribution data of mobile species such as deep-sea fish are poor, with many migration routes and aggregation areas currently unknown (Haedrich et al., 2001). Without sound scientific knowledge, detailed species distributional and abundance data and physical habitat mapping, it is difficult to answer the three essential questions for effective reserve design posed by Carr et al. (2003).

One suggested solution is the development of a marine reserve network extending throughout coastal areas and the High Seas (Roberts et al., 2006a). Networks of protected areas could protect highly migratory species, and may even protect undiscovered habitats such as those associated with seamounts (Roberts et al., 2006a). Another potential mechanism to protect migratory deep-sea species could be mobile reserves that would follow sensitive species along migration routes (Roberts et al., 2006a). In the deep sea, potentially the most immediately effective measure would be to allow fishing to continue in areas where fish stocks have reduced and benthic damage already occurred but to close other areas to new fishing to protect existing fish stocks and benthic habitats. However, this would need to be coupled with effort reductions in order to protect existing fish stocks which are rapidly diminishing (Baum et al., 2003; Morato et al., 2006). It is likely that any biologically sustainable deep-sea fish catches would be economically worthless.

There is still much to learn about the deep sea, making comparisons with existing coastal protected areas difficult. Coastal reserves are found within the exclusive economic zones of nations allowing the establishment of effective legalisation. Reserves in coastal areas often have significantly higher organism density, biomass and diversity than comparative control areas (Murawski et al., 2000; Halpern, 2003). It is likely the recovery of deep-sea habitats and fish stocks would occur, providing reserves are effectively designed and policed. Recovery would probably take longer, especially as many deep-sea species have slower growth rates, later sexual maturation and variable or infrequent recruitment, but there has been no research recording the effectiveness of deep-sea marine protected areas. There are many challenges surrounding the designation of protected status for deep-sea areas, but there could be extensive economic and social benefits for doing so, as outlined for shallow-water marine protected areas by Balmford et al. (2002, 2004).

### 3.1. Protection initiatives

Throughout the world, there are over 100 000 designated protected areas covering a total area of 18.8 million km<sup>2</sup>. Of these, marine environments constitute 1.64 million km<sup>2</sup>, representing less than 0.5% of the global ocean surface (Chape et al., 2003). To date, only few deep-sea areas have been granted permanent protected status, often these areas contain focal species and habitats such as cold-water corals (See Table 1 and Freiwald et al., 2004) and hydrothermal vents (e.g. Endeavour Hydrothermal Vent Field, Canada).



**Table 1 – Current established cold-water coral protected areas throughout the world**

Country/Area	Year of designation	Cold-water coral protection (Restrictions on bottom contact fishing gear/human activity)
Australia	1999	Tasmanian Seamount Reserve. Seamounts ( <a href="#">Commonwealth of Australia, 2002</a> ).
Azores, Madeira and Canary Islands	2004	Deep-water coral reefs ( <a href="#">European Council, 2004</a> )
Canada	2002	Northeast Channel Conservation Area. Deep-water coral ( <a href="#">Department of Fisheries and Oceans, 2002</a> )
	2004	Stone Fence Fisheries Closure. Cold-water coral reefs ( <a href="#">Department of Fisheries and Oceans, 2004</a> )
	2004	The Gully Marine Protected Area. Protection for all species including corals ( <a href="#">Canada's Oceans Act, 2004</a> )
Europe	2006	Capo Santa Maria di Leuca. <i>Lophelia</i> reefs ( <a href="#">FAO GFCM, 2006</a> )
New Zealand	2001	Seamounts management strategy, seamount fauna including corals ( <a href="#">Brodie and Clark, 2004</a> )
Norway	1999	Sula Reef. Cold-water coral reefs ( <a href="#">Norwegian Directorate of Fisheries, 2006</a> )
	2000	Iverryggen. Cold-water coral reefs ( <a href="#">Norwegian Directorate of Fisheries, 2006</a> )
	2003	Røst Reef. Cold-water coral reefs ( <a href="#">Norwegian Directorate of Fisheries, 2006</a> )
	2003	Tisler Reef. Cold-water coral reefs ( <a href="#">Norwegian Directorate of Fisheries, 2006</a> )
	2003	Fiellknausene. Cold-water coral reefs ( <a href="#">Norwegian Directorate of Fisheries, 2006</a> )
USA	1984	Oculina protected area. <i>Oculina</i> reefs, Florida. ( <a href="#">South Atlantic Fishery Management Council, 1998</a> )
	2005	Aleutian Islands. Deep-sea corals. ( <a href="#">Department of Commerce, 2006</a> )
UK	2004	Darwin Mounds, coral mounds ( <a href="#">European Council, 2004</a> )

Some areas have been granted temporary protection against potentially damaging activities such as fishing, for example the NEAFC placed a 2-year restriction on fishing gear type at five Mid-Atlantic Ridge Seamounts on the 1st January 2005 ([NEAFC, 2004](#)). Unlike these complex habitats, the majority of the deep sea appears a relatively featureless expanse of sediment, lacking in focal species and habitats. This does not mean these areas are of lower conservation value; rather, some estimates place them amongst the most diverse habitats on earth ([Grassle and Maciolek, 1992](#); [Gage, 1996](#)). The focal species approach is potentially a powerful tool for conservation ([King and Beazley, 2005](#)). The rationale is that focal species may warrant conservation because they possess characteristics that identify them as functionally important, vulnerable or sensitive to disturbance.

Cold-water corals meet many of these criteria ([King and Beazley, 2005](#)) and are now receiving considerable conservation focus (see [Freiwald et al., 2004](#); [Morgan et al., 2006](#)). The immediate protection of sensitive areas such as cold-water coral ecosystems may be a first step to developing a network of protected areas throughout the oceans. Over the last few years, bottom trawling has been identified as one of the most significant threats facing cold-water corals ([Hall-Spencer et al., 2002](#); [Gage et al., 2005](#); [Grehan et al., 2005](#); [Roberts et al., 2006b](#); [Wheeler et al., 2005](#)). This has led to significant publicity and lobbying to ban bottom trawling in areas of coral habitat. In late 2006, United Nations General Assembly called upon member states and regional fisheries management organisations to close areas to bottom fishing in areas where seamounts, hydrothermal vents and cold-water corals are known or are likely to occur based on scientific information (UN GA, Draft resolution of the 61st session of the General Assembly, 6th December 2006, A/61/L.38). This represents one of the first major international steps to protect vulnerable habitats in the deep sea.

### 3.2. Case study: conserving cold-water corals

Cold-water corals are an important flagship species for conservation in the deep sea ([King and Beazley, 2005](#)) that are

becoming relatively well understood compared to other deep-sea habitats ([Freiwald et al., 2004](#); [Roberts et al., 2006b](#)). Reef and mound features are diverse habitats. For example, one study examining 11 box-cores from carbonate mounds in the Porcupine Seabight reported 10 previously undescribed benthic invertebrates and found coral-rich cores from the mounds to be three times more species-rich than cores taken off mound ([Henry and Roberts, 2007](#)). Several countries are now actively protecting cold-water coral areas within their national waters ([Table 1](#)). Further offshore, the High Seas also contain many vulnerable habitats and new regulations are required to protect areas that lie in international waters. Currently, there is only one example of a permanently protected area in a region fished by several European Union member states and other licensed states.

The Darwin Mounds lie 180 km off the north-west of Scotland at ca. 900 m water-depth within the European Union Exclusive Economic Zone ([Bett, 2001](#)). These mounds are colonised by a diverse community of corals, suspension feeders and infauna ([Masson et al., 2003](#); [Van Gaever et al., 2004](#)). Discovered in 1998, the mounds were designated an offshore marine protected area in 2004 after concerns about possible damage from hydrocarbon exploration and proven damage from bottom trawling ([Wheeler et al., 2005](#)). In 1999, Greenpeace campaigned to halt offshore oil and gas exploration to the west of the UK, leading to a High Court judgment that the European Community Habitats Directive is applicable throughout the UK's continental shelf and any waters within the 200 nautical mile limit of member states exclusive economic zones or exclusive fishing zones. Under the Habitats Directive, member states are required to take measures to maintain or restore certain specified natural habitats and wild species by assuring their protection. Reports concerning the increasing extent of trawling damage to seabed habitats at 1000 m depth on the European continental margin ([Roberts et al., 2000](#); [Hall-Spencer et al., 2002](#)) and the detection of trawl marks across the Darwin Mounds in 2000 ([Wheeler et al., 2005](#)) led to growing calls for this area to be protected from further damage by prohibiting bottom trawling within the Darwin Mounds area. In 2003 this led to a formal request

from the UK government to the European Commission for an emergency closure of the Darwin Mounds under the European Common Fisheries Policy (European Council, 2003a). The following year, the European Council announced that from August of 2004 an area of 1300 km<sup>2</sup> around the Darwin Mounds would be permanently closed to bottom trawling (European Council, 2004) and the UK plans to designate the area as a special area of conservation (Annex 1, EC Habitats Directive) once enabling legalisation is in place.

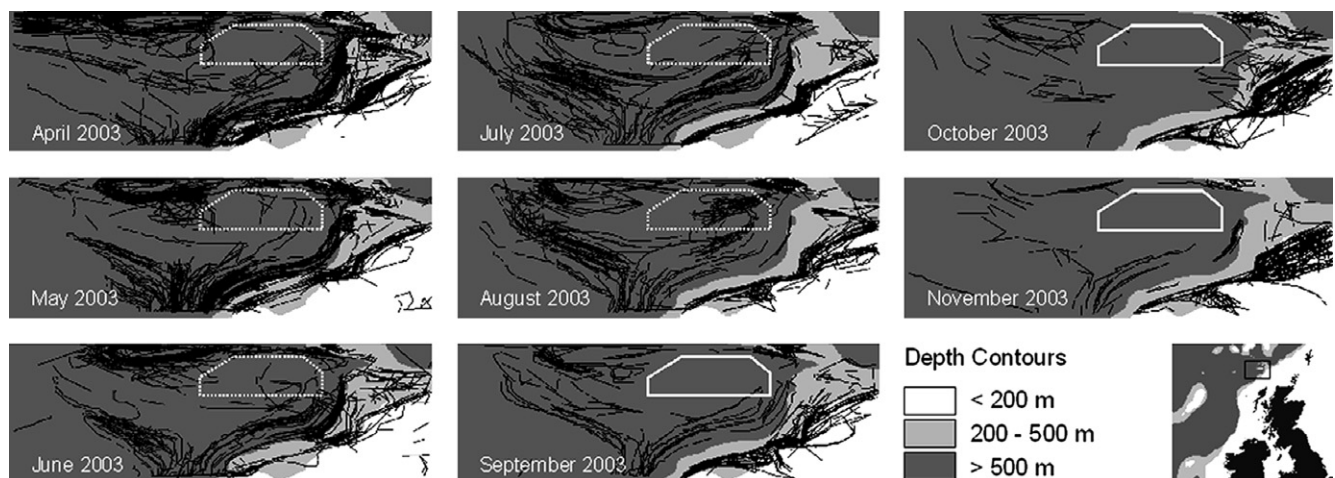
### 3.3. Enforcement of protected status

It has become increasingly important that measures are developed to effectively manage and police deep-sea protected areas. Many, if not most, such areas are found miles offshore with no visual barriers or landmarks. In coastal areas, spotter planes, patrol vessels and onboard observers are often used to monitor protected areas. However, applying these methodologies to deep-sea monitoring may be prohibitively expensive and time-consuming, offering only limited spatial coverage. Some deep-sea fisheries use observer systems, such as Canadian vessels fishing for Greenland Halibut and the Falkland Island fisheries. One cost effective method is the emerging use of global positioning surveillance (Marr and Hall-Spencer, 2002; Deng et al., 2005; Murawski et al., 2005) or remotely sensed images (Kourti et al., 2001, 2005). These methodologies offer complete spatial coverage and may become increasingly cost effective in the long term.

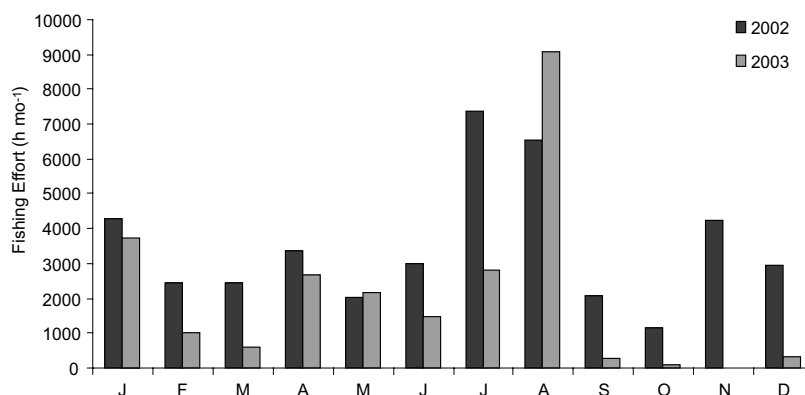
Since 2002 European Union fishing vessels greater than 24 m in length (reduced to all vessels greater than 15 m in length since 2005) have been required to submit their position via GPS every two hours to a Fisheries Monitoring Centre (European Council, 2003b). Each European Union member state receives Vessel Monitoring System (VMS) data for vessels that are active within its exclusive economic zone and the global positions of vessels that are registered to that member state. At present these data are not used for deep-

sea habitat protection, but they could be used as an additional tool for the spatial management of offshore fishing fleets (Deng et al., 2005; Murawski et al., 2005). There are, however, problems associated with this approach. There is often uncertainty over the type of fishing taking place which requires corroborative evidence such as visual sightings in closed areas (e.g. Kuruc, 2005) and VMS can be blocked or falsified, which is leading some government authorities to investigate the use of remotely sensed imagery as a complementary tool to support VMS evidence (Kourti et al., 2005).

The potential for VMS as a fisheries management tool in the deep sea can be illustrated using the Darwin Mounds closed area. Vessel monitoring system data collected during 2003 for fishing vessels >24 m in length was supplied by the UK Department of Environment Food and Rural Affairs. The data were from all EU countries, including the Faeroe Islands and Norway and was filtered to retain the tracks of vessels travelling between 1.5 and 4.5 knots as deep-water trawlers cannot fish outside these speeds (J.D.M. Gordon, pers. comm.). Vessels travelling at trawling speeds were most active on the banks and slopes surrounding the Darwin Mounds province, with little fishing activity in the closed area (Fig. 3). There was a large increase in fishing activity recorded in the month before the closure (Fig. 3). It is possible that the announcement of the closure may have directed increased trawl effort into the Darwin Mounds protected area, with fishing effort (hours spent trawling) for 2003 increasing between July and August (Fig. 4). However, this conclusion may be misleading as comparison with data from 2002 shows a similar seasonal increase in fishing effort in August within the closed area (Fig. 4). Whilst there is no evidence for a reduction in overall effort throughout the overall area, the potential for VMS data as a monitoring and enforcement tool is great; it can graphically demonstrate breaches of area closures and could help to warn fishermen that they are about to enter a restricted area. Thus, VMS may become one of the most valuable tools available to monitor High Seas marine protected areas and assess fishing activity.



**Fig. 3** – Monthly tracks of fishing vessels travelling at typical trawling speeds of 1.5–4.5 knots derived from VMS data for the Darwin Mounds box for 2003. The closed area is indicated by a white box (dashed line, pre-closure; solid line, post-closure). Inset map shows the location of the Darwin Mounds (white box). Bathymetry from Gebco (IOC et al., 2003).



**Fig. 4 – Fishing vessel effort (hours per month) within the Darwin Mounds area that was closed to trawling in September 2003 based on VMS data. This provides a minimum estimate of effort since vessels were assumed to be (a) travelling in straight line between position fixes that were received every 15–120 min and (b) trawling when travelling at speeds of 1.5–4.5 knots.**

#### 4. Concluding comments

As our understanding of deep-sea habitats improves, many more threats to these environments are uncovered. In recent years, bottom trawl fishing has been highlighted as one of the most significant, historical over-fishing and serial depletion of stocks has led to large collapses in fish stocks world-wide. Trawling is a well-recognised cause of habitat damage in deep-waters, especially to benthic ecosystems dominated by sessile corals and sponges. The ever-increasing search for new hydrocarbon resources and the emerging potential for CO<sub>2</sub> storage in the deep-sea may have significant effects on deep-sea communities. On a global scale, climate change is becoming the most significant impact with a reach predicted to extend to the deepest parts of the ocean.

Successful management, protection and restoration of deep-sea habitats require a sound scientific basis, which is currently lacking. Increased mapping and basic ecological research is needed to understand the distribution and dispersal capabilities of deep-sea organisms which will assist in the design of marine reserves. Current protection, although improving, still needs to become more efficient with comprehensive legislation and enforcement that can cover multi-jurisdictional areas such as the High Seas. Conservation measures must be monitored and enforced. The vessel monitoring system shows great potential as a tool to monitor fishing activities offshore and on the High Seas. The EU has identified the year 2012, as the target date to develop an ecologically coherent network of marine reserves. With so much of the earth affected by human activity, it is important to ensure that the deep sea, very probably the last great wilderness on earth, attains adequate protection to preserve it for future generations.

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