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THE NEAR FUTURE OF THE DEEP SEAFLOOR ECOSYSTEMS

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SUMMARY

The deep-sea floor is vast, and contains a broad array of habitats: these include sediment-covered slopes and plains, rocky mid-ocean ridges and seamounts, and island-like chemoautotrophic communities ranging from hydrothermal vents to whale falls. Communities in many of these habitats are likely to be very susceptible to anthropogenic disturbance due to low rates of productivity, growth and colonization, and delicate habitat structure. A small subset of deep-sea communities, in particular those at hydrothermal vents, may be robust in the face of human impacts because of natural physical dynamism, and rapid growth rates and broad dispersal abilities of the resident fauna. Although the deep-sea appears remote from human activities, a variety of anthropogenic forcing factors already impact deep seafloor communities, and these impacts will increase in the near future. The effects of fishing are perhaps most prevalent, with a number of seamount and continental-slope fish stocks already exploited to commercial extinction. In addition, the highly diverse and potentially endemic assemblages living on seamounts and in deep-sea coral beds are severely threatened by the physical devastation caused by bottom trawling. Oil and gas exploitation is increasing dramatically on deep continental slopes to water depths of at least 2000 m, and significant environmental impacts resulting from the release of drill cuttings are to be expected in the near future. The mining of other geological resources from the seafloor, in particular manganese nodules, is probably a few decades off but has the potential to negatively impact large seafloor areas (10,000 to >100,000 km^2) through “strip mining” of nodules and redeposition of suspended sediments. The potential environmental risks posed by nodule mining, in particular the chance of species extinctions, cannot be evaluated because of limited knowledge of colonization rates and species ranges on the abyssal plains. Waste disposal, particularly of sewage sludge and dredge spoils, has occurred in the past on continental slopes, and is likely to increase in the near future as terrestrial disposal options become more limited. The release of sewage sludge and dredge spoils at the ocean surface has impacted seafloor communities over large areas, although the intensity of effects documented thus far appear to be modest. Climate change will inevitably alter the deep-sea benthos because deep-sea communities are tightly linked to production processes in the surface

ocean. The few long time-series studies of deep-sea assemblages suggest climate-induced community changes may already be taking place. Pollutant loading (e.g., of PCB’s, mercury and cadmium) appears to be widespread in deep-sea animals, and some species may already experience toxic effects. Despite substantial and increasing anthropogenic impacts in the deep sea, deep seafloor ecosystems are so poorly understood that the impacts of human activities cannot be effectively predicted or managed. Research is required to elucidate several key topics, including: (1) species ranges, rates of gene flow, and the distribution of critical seafloor habitats at the deep-sea floor; (2) the response of deep-sea ecosystems to particular anthropogenic stressors, especially trawling, oil drilling, mining, waste disposal, and pollutants; (3) vulnerable deep-sea species and ecological processes, as well as useful indicators of ecosystem health; and (4) natural variability in a range of deep-sea ecosystems over time scales of years to decades. In conclusion, anthropogenic impacts are increasing rapidly in the deep sea while the ecological study of deep-sea habitats has lagged far behind that of other ecosystems. This state of affairs, combined with the complex nature of conservation management on the “high seas”, means that deep-sea conservation requires the urgent attention of research-funding bodies and policy makers if we are to preserve the biological wonders of the Earth’s last real frontier.

I. INTRODUCTION

The deep-sea floor lies between the shelf break (~ 200 m depth) and the bottom of the Challenger Deep (~11,000 m). It is staggeringly vast, spanning more than 300 x 10^6 km^2, or approximately 63% of the Earth’s solid surface. The distinct habitats of the deep seafloor are varied and include sediment-covered slopes, abyssal plains, and ocean trenches, the pillow basalts of mid-ocean ridges, rocky seamounts protruding above the seafloor, and submarine canyons dissecting continental slopes. The sedimented plains of the slope and abyss are the largest in area, covering > 90% the deep-sea floor and often extending unbroken for over a thousand kilometers. Deep-sea trenches, where continental plates overrun oceanic crust, constitute 1-2% of the deep-ocean bottom. The rocky substrates of mid-ocean ridges (ribbons ~10 km wide and, in total, ~60,000 km long), seamounts (perhaps 50,000 – 100,000 in number (Epp & Smoot, 1989; Smith, 1991; Rogers, 1994)), and submarine canyons are relatively rare habitats in the enormous expanses of the deep sea, together estimated to occupy < 4% of the seafloor.

Many deep seafloor habitats share ecological characteristics that make them especially sensitive to environmental change and human impacts. Perhaps the most important characteristic is low biological productivity. Away from the occasional hydrothermal vent and cold seep, the energy for the deep-sea biota is ultimately derived from an attenuated "rain" of organic matter from surface waters 100's to 1000's of meters above (typically 1 – 10 g C_{org} m^{-2} y^{-1}). Detrital particles range in size from phytoplankton cells a few micrometers in diameter to dead whales 30 m long. The purely detrital base of most deep-sea food webs contrasts sharply with those of other marine and terrestrial ecosystems, which typically are sustained by local production (Polunin *et al.*, 2001). The very low organic-energy flux, combined with low temperatures (-1 – 4°C) in the deep sea, yields communities generally low in biomass and characterized by relatively low rates of growth, respiration, reproduction, recolonization and sediment mixing (Gage & Tyler, 1991; Smith & Demopoulos, 2003). In addition, many deep-sea animals, especially macrofaunal polychaetes, molluscs and crustaceans, are very small in body size and extremely delicate (Gage & Tyler, 1991; Smith & Demopoulos, 2003).

The deep-sea floor is also generally characterized by very low physical energy, including sluggish bottom currents (typically < 0.25 knots), very low sediment accumulation rates (0.1 – 10 cm per thousand years), and an absence of sunlight (Gage & Tyler, 1991; Smith & Demopoulos, 2003). Nonetheless, the seemingly monotonous sediment plains of the deep sea often harbor communities with very high local species diversity, with one square meter of deep-sea mud containing hundreds of species polychaetes, crustaceans and mollusks (Snelgrove & Smith, 2002).

However, not all deep-sea habitats are low in energy and productivity. Hydrothermal vents, and to a lesser degree, cold seeps, may sustain high productivity, animal growth rates, and community biomass as a consequence of chemoautotrophic production by abundant microbial assemblages fueled by reduced chemicals such as hydrogen sulfide (Van Dover, 2000). Seamounts, canyons, whale falls, and upwelling zones may also foster relatively high biomass communities by enhancing bottom currents or by concentrating organic-matter flux (Wishner *et al.*, 1990, Koslow, 1997; Vetter & Dayton, 1998; Smith & Baco, 2003; Smith & Demopoulos, 2003).

The sensitivity and recovery potential of deep-sea ecosystems in the face of human disturbance are not necessarily easy to predict. The relatively low levels of physical energy, productivity and biological rates, combined with small body size, suggest that most deep-sea habitats are especially sensitive to, and slow to recover from, deleterious human impacts compared to most other ecosystems (Table 1). The relatively high species diversity in the deep sea, in terms of the number of species per unit area, again may make the deep-sea habitats more sensitive to human impacts (there are more species to be extinguished by local disturbance, and more species that must recolonize). Yet the size of the ecosystem is especially large, and for abyssal soft-sediments, habitats are nearly continuous across ocean basins. The large habitats of the deep-sea may make the fauna more resistant to extinctions caused by local perturbations, with a greater potential for recolonization from widespread source populations. However, the size of source populations depends to a large extent on the biogeographic ranges of deep-sea species, which are very poorly known (Glover *et al.*, 2002). Large, continuous habitats may also allow stressors, such as disease agents, toxic chemicals or radioactive contaminants, to disperse over vast distances, and to become amplified over very large areas as they move up deep-sea food webs. Clearly, the unusual characteristics of deep-sea ecosystems present novel challenges to the prediction of anthropogenic impacts and requisite conservation actions.

While once considered remote and well buffered from the euphotic zone and from human impacts, deep-sea ecosystems are increasingly recognized to be linked to processes in the upper ocean. In fact, until the early 1980's, it was widely believed that the deep-sea floor was a remote and deliberate habitat, where low current speeds, low temperatures and a gentle rain of organic material drove biological processes at extremely slow, relatively constant, rates (Smith, 1994). However, more recent data from deep sediment traps and time-series studies in the Atlantic and Pacific show marked temporal variability in particulate organic flux, accumulation of fresh phytodetritus, sediment community respiration rates, and reproductive patterns at the abyssal seafloor (Deuser &

Ross, 1980; Tyler *et al.*, 1982; Tyler, 1988; Thiel *et al.*, 1989; Gage & Tyler, 1991; Smith *et al.*, 1996; Smith & Kaufmann, 1999; add 2004 KLS ref). In addition, the abundance and biomass of deep-sea benthos appear to be strongly correlated with large-scale spatial and temporal variations in export production from the euphotic zone (Smith *et al.*, 1997; Smith *et al.*, 2001; Smith and Demopoulos, 2003). Clearly, changes in upper-ocean ecosystems resulting from climate change or direct human activities such as overfishing (Myers & Worm, 2003) are bound to alter the structure and functioning of deep seafloor ecosystems.

Why should we care whether the seemingly remote habitats of the ocean bottom are altered by human activities in the near future? There are a number of compelling reasons. (1) The deep-sea floor is one of Earth's the largest ecosystems; we can hardly assess the ecological "health" of our planet without considering its vastest habitats. (2) The deep-sea floor is both an extreme environment and a substantial reservoir of biodiversity (Snelgrove & Smith, 2002). Its biota offers exciting insights into evolutionary novelty as well unusual biotechnological resources. (3) The floor of the ocean provides a number of ecosystem services fundamental to global geochemical cycles. For example, deep-sea sediments are major sites of nutrient recycling, and are responsible for much of the carbon burial occurring on the planet. (4) Because deep-sea ecosystems are significantly buffered from the high-frequency variations in upper-ocean processes occurring on short space and time scales, ecosystem parameters in the deep-sea may act as "low-pass filters," primarily responding to gradual, large-scale changes in production processes in the euphotic zone (e.g. Hannides & Smith, 2003). Thus, deep-sea ecosystems may be especially useful in elucidating basin-scale trends in ocean productivity driven by global climate change.

In the following sections, we discuss existing anthropogenic alterations to deep-sea ecosystems, and additional changes expected to occur in the near future (i.e., over the next 1-2 decades). The anthropogenic forcing factors we consider include fishing, energy and mineral exploitation, waste disposal, climate change, and general (i.e., "non-point-source") pollution. We then discuss the research needed to elucidate such changes, and discuss the formulation of policies and regulations that might contribute to the mitigation of deleterious anthropogenic impacts. Finally, we draw conclusions about what the near future holds for deep-sea ecosystems, and what humankind might want to do about it.

II. NATURAL AND ANTHROPOGENIC FORCING FACTORS

A. Effects of Deepwater Fisheries

Ecology and life-history patterns. Since World War II, global fishing activity has increasingly targeted deepwater species on the continental slope as well as on seamounts, banks and plateaus deeper than ~500 m. In part, this fisheries expansion was made possible by technological developments, such as of echo sounders, GPS, net sondes, track plotters, and rock-hopper trawls. This expansion was driven as well by the “fishing-up” process, whereby fisheries increasingly exploit less accessible and more marginal fishing grounds, often containing less desirable species, following the depletion of traditional fisheries (Deimling & Liss, 1994). Deepwater fishing activities, particularly those carried out by demersal trawling, have probably had a greater ecological impact than any other human activity in the deep sea.

Deepwater fisheries generally target different families of fishes, and often different orders, than traditional fisheries. These newly exploited groups of fishes generally exhibit markedly different life histories, and the ecological characteristics of the communities they inhabit differ as well. As a result, there is reason to believe that the direct and indirect impacts of many deepwater fisheries will be far more intense and persistent than those of traditional shallow-water fisheries, despite the relatively brief period that deepwater stocks have been exploited. In fact, a marked “boom and bust” cycle already characterizes deepwater fisheries: most have been massively depleted within ten years of inception (Koslow *et al.*, 2000).

The species exploited by deepwater fisheries vary considerably depending upon habitat (e.g., deep banks and seamounts versus the continental slope) and also depending upon the ocean basin and biogeographic province. Seamount populations exhibit reproductive isolation across a range of spatial scales, such that different orders of fishes evolved to dominate seamounts in different ocean basins and climatic regions. For example, pelagic armourhead (*Pseudopentaceros wheeleri*) (Pentacerotidae, Perciformes) dominate seamounts in the central North Pacific; several species of *Sebastes* (Scorpaenidae, Scorpaeniformes) prevail along the continental slope of the North Pacific and North Atlantic; orange roughy (*Hoplostethus atlanticus*) (Trachichthyidae, Beryciformes) and several oreosomatid species (Zeiformes) dominate slopes and seamounts in the temperate South Pacific; and alfonsino (*Beryx* spp.) (Berycidae, Beryciformes) abound in the tropics and sub-tropics (Koslow *et al.*, 2000). Many, although not all, of these species are exceptionally long-lived. Orange roughy, and some oreosomatids and species of *Sebastes*, live more than 100 years (Smith & Stewart, 1994; Tracey & Horn, 1999; Caillet *et al.*, 2001) and thus must experience extremely low natural mortality (≤ 0.05) and slow post-maturity growth. There are few time-series studies of recruitment for deepwater species, but available data indicate life histories characterized by extreme iteroparity over a prolonged reproductive period punctuated by occasional episodes of recruitment (Murphy, 1968; Stearns, 1976). For example, populations of orange roughy and *Sebastes* may go for a decade or more with very low recruitment (Leaman & Beamish, 1984; Clark, 1995).

Deepwater fisheries on the continental slope predominantly target several families of Gadiformes: grenadiers or rattails (Macrouridae), morid cods (Moridae), hakes (Merluciidae) and cusk eels (Brotulidae). Other key groups include flatfishes within the Pleuronectidae (e.g. Greenland halibut, *Reinhardtius hippoglossoides*, one of the few families fished both on the continental shelf and in deep water), and various species of *Sebastes*. Species in these groups tend to be long-lived (50-100 years) but don't appear to exhibit the extreme ages of seamount-associated species (Bergstad, 1990; Campana *et al.*, 1990).

Fisheries development and direct impacts. The low productivity and episodic recruitment of deepwater species renders them highly vulnerable to over-exploitation. Since the 1960s, deepwater fisheries landings have fluctuated between 600,000 and 1 million metric tonnes, but underlying this apparent stability is a pattern of serial depletion (Koslow *et al.*, 2000) (Fig. 1). Modern deepwater fisheries originated in the North Pacific and North Atlantic. Several species of *Sebastes,* e.g. *S. alutus* in the Pacific and the redfish complex (*S. marinus, S. mentella and S. fasciatus*) in the North Atlantic, were first fished along the shelf edge and upper slope and are included among the deepwater fisheries because they share many life-history characteristics with deepwater species. These have been the largest and most stable deepwater fisheries, but Koslow *et al.* (2000) conclude: “uncertainties about the unit stocks, and in the North Atlantic even the taxonomic composition of the catch, appear to have masked the progressive fish down of successive populations around the rim of both the North Pacific and the Atlantic.”

Coryphaenoides is a diverse genus of macrourids, dominant over relatively flat portions of the deep sea in much of the world’s oceans. There has been a substantial but diminishing fishery for *C. rupestris* in the North Atlantic and a small fishery for *C. acrolepis* in the North Pacific (Merrett & Haedrich, 1997). *Coryphaenoides rupestris* is most abundant between 600-800 m depths. The Soviet fleet developed the fishery in the northwest Atlantic, which peaked at 80,000 tonnes in 1971 and then rapidly declined; landings in 1997 were only a few hundred tones. These fisheries, which extend out into international waters along the mid-Atlantic ridge, are not actively managed.

The “boom and bust” pattern is most marked for seamount-associated species, which aggregate on fixed topographic features and generally exhibit the most extreme life-history characteristics. Given the low productivity of these stocks, the fisheries tend to be unsustainable: A large accumulated biomass is fished down, usually within ten years, and the fishery moves on to another area. The first seamount fishery targeted pelagic armourhead in the central North Pacific in the 1970s. Within about 10 years of their discovery by Soviet trawlers, the population was fished to commercial extinction (Boehlert, 1986). A new species of *Corallium* (a precious red coral) was discovered and largely fished out on seamounts in the North Pacific in the 1980s (Grigg, 1993).

Soviet trawlers discovered orange roughy on seamounts around New Zealand in the 1970’s. In the 1980’s and 1990’s, orange-roughy stocks were successively depleted around southeastern Australia and New Zealand, followed by Namibia, and most recently in the Indian Ocean (Clark, 1999; Koslow *et al.*, 2000). Australia and New Zealand intensively managed their larger orange roughy stocks, but with limited success due to the low resilience of the stocks to overfishing (Clark, 1999; Bax, 2000); several key New Zealand orange-roughy fisheries have now been closed due to decline to only a few percent of virgin biomass. Smaller stocks, and stocks in international waters, are generally left unmanaged; fishing ceases only when it is no longer commercially viable.

The global depletion of deepwater fish species has been too recent to assess long-term impacts. The ability of these stocks to recover is not known, but, given their late maturity and sporadic recruitment, it will at best be very slow compared to most shallow-water fish species.

Indirect fisheries impacts. In addition to reducing target-species population sizes, fisheries may have pervasive secondary impacts on the target species, including density-dependent changes in growth, reproduction and recruitment. In addition, fisheries may dramatically impact other components of the ecosystem, altering the size structure, species composition and food webs of benthic communities due to bycatch and predator removal (Jennings & Kaiser, 1998; Hall, 1999). Trawl-based fisheries in particular may also cause substantial physical disruption of seafloor habitats by plowing sediments and fragmenting the biogenic structure of corals, sponges, gorgonians, etc. (Jennings & Kaiser, 1998; Hall, 1999; Koslow *et al.*, 2000). The large-scale removal of seabirds, top predators and fishes, as well as the physical disruption of the seafloor, has raised the specter of massive and possibly irreversible fishing down and removal of habitat complexity from marine ecosystems (Sainsbury, 1988; Collie *et al.*, 1997; Pauly *et al.*, 1998; Thrush & Dayton, 2002; Myers & Worm, 2003)

Fishery impacts in the deep sea are generally poorly known. Fishing on the high seas is *ipso facto* unregulated and poorly documented, and research on deepwater habitats and communities presents obvious challenges and greater cost. On the other hand, the recent development of many deepwater fisheries, coming at a time of greater awareness of fishery impacts, has enabled studies to be made for some deepwater fisheries at an early stage, which was not possible for many traditional fisheries that began decades or even centuries earlier. In general, those fishery-induced changes observed in populations and ecosystems on continental shelves also appear to be occurring in deepwater habitats.

Fishing appears to have induced density-dependent changes in some, though not all, target populations studied. The fecundity of the Tasmanian orange-roughy population increased 20% following a 50% reduction in stock size due to fishing (Koslow *et al.*, 1995), but a similar effect was not observed for the Challenger Plateau (New Zealand) stock of orange roughy (Clark *et al.*, 1994). Enhanced growth was observed in fished populations of *Sebastes alutus* and *S. diploproa* (Boehlert *et al.*, 1989). The impact of reduced spawning stock size on recruitment to deepwater populations has not been examined due to a lack of suitable recruitment time series.

Fishing has reduced the size structure of deepwater fish assemblages in the North Atlantic (Haedrich, 1995; Large *et al.*, 1998). Perhaps surprisingly, trawl surveys in the North Atlantic (Lorance, 1998) and of New Zealand orange-roughy fishing grounds (Clark & Tracey, 1994; Clark, 1995) indicated no major shifts in species composition: however, this was because the relatively non-selective nature of the trawl fisheries reduced the abundance of most species to simultaneously.

The impacts of deepwater fisheries on non-target populations, including demersal fish and hard-bottom benthos, are a major concern. For example, the take of of non-targetted species (or bycatch), is considered responsible for 80-97% reductions over a 17 yr period in the population densities of the demersal fishes *Antimora rostrata*, *Bathyraja spincauda* and *Notocanthus chemnitzi* at 200-2500 m on the North Atlantic slope (Baker and Haedrich, unpublished data). In addition, deepwater trawl fisheries often target topographic features (seamounts, banks, canyons) with enhanced productivity; such features frequently also harbor distinct benthic communities dominated by hard and soft corals, sponges and other suspension feeders. *Lophelia* reefs in the North Atlantic and deepwater coral reefs in the South Pacific can be tens of meters high and represent millennia of growth (Rogers, 1999). The biodiversity of these reefs rivals that of shallow-water tropical reefs; seamounts also exhibit high levels of endemism, suggesting little genetic exchange between seamount chains (de Forges *et al.*, 2000). Coral bycatch is reported from deepwater trawl fisheries as new fishing grounds are entered (Probert *et al.*, 1997; Anderson & Clark, 2003). Massive damage to deepwater coral habitats has been reported from fisheries off Tasmania, the European continental margin, and North America (Koslow *et al.*, 2001; Krieger, 2001; Fossa *et al.*, 2002; Mortensen *et al.*, in press). Recolonization processes for deepwater corals are unknown. Given the slow growth of deepwater corals (e.g. 5.5 mm/yr for *Lophelia pertusa* (Rogers, 1999)) and uncertain rates of recruitment, the re-establishment of deepwater coral reefs will likely take centuries to millennia. As a result, deepwater marine protected areas have been declared off Tasmania, New Zealand, Norway and elsewhere (Koslow *et al.*, 2000). Recognizing the grave environmental consequences of high-seas trawl fisheries, the UN General Assembly passed a resolution in 2002 to protect the biodiversity of the high seas. The extent of potentially irreversible damage already inflicted to seamount and deepwater coral habitats is unknown.

B. Energy and Mineral Exploitation

Oil and Gas Exploitation. In the 1990's, the offshore oil industry expanded exploratory drilling and production into the deep sea below 500 m, with the Brazilian oil company Petrobras now working at water depths >2000 m. Very large oil reserves have been discovered in water depths exceeding 1000 m in the Gulf of Mexico and off West Africa (Douglas-Westwood, 2002) suggesting that deep-sea oil production will develop substantially in the next decade.

The environmental effects of oil and gas drilling are reasonably well documented at shelf depths, and should be similar, albeit with higher disturbance sensitivities and slower recovery times, in the deep sea. On the shelf, principal benthic impacts result from release of drilling muds and drill cuttings. Drilling muds can include refined lubricant oils and other synthetic components, and are used to lubricate the drill bit and carry drill cuttings out of the well. Drill cuttings and drilling mud are separated on the drill platform and the cuttings discharged back into the sea, where they may accumulate on the seabed. The piles of cuttings are usually contaminated with drilling muds, and pose a significant risk to marine life (Daan & Mulder, 1996; Raimondi *et al.*, 1997; Mauri *et al.*, 1998; Grant & Briggs, 2002).

Large quantities of drill cuttings have accumulated around drilling platforms; for example, 30 years of drilling in the North Sea has left between 1 and 1.5 million tonnes of drill cuttings on the seafloor (UKOOA, 2002). The principal impacts of drill cuttings on the seabed include physical smothering, organic enrichment and chemical contamination (by hydrocarbons, heavy metals, special chemicals and sulphides) of the benthos near the cuttings source (Daan & Mulder, 1996; Mauri *et al.*, 1998; Grant &

Briggs, 2002). Experimental studies indicate that drilling muds can also inhibit the settlement of marine invertebrate larvae (Raimondi *et al.*, 1997). Deep-sea drilling also includes the risk of catastrophic failures in offshore operations (e.g., the explosion and sinking of the giant drilling rig “P-36” in 2000 m of water off the Brazilian coast in 2001). The environmental impact of such disasters will depend on the frequency of their occurrence.

When drill cuttings are deposited in shelf areas with high current speeds, the fine rock chips are usually dispersed quickly, allowing biodegradation of drilling-mud contaminants. In the northern North Sea, where water is deeper and current speeds are lower, drill cuttings have accumulated, causing significant localized environmental impacts (UKOOA, 2002). It very likely that drill cuttings will cause even greater local environmental hazard in the deep sea, where current speeds, sediment accumulation rates, and recolonization rates often are very low, yielding relatively low community resistance to burial and slow community recovery. In addition, the low background levels of productivity and biomass will make the impact of any organic enrichment more significant.

Forecasting the potential environmental impacts of deep-water drill cuttings is problematic given the uncertainties associated with future oil discoveries. Studies from the Norwegian shelf indicate that a single oil rig may impact a seafloor area of roughly 100 km^2 (Olsgard & Gray, 1995). Currently, the most active region of deepwater drilling in the world appears to be the Gulf of Mexico, where roughly 50 drilling rigs are in operation in waters deeper than 500 m; if each rig has an ‘impact zone’ of 100 km^2, this amounts to a total of 5000 km^2 of deep-sea impact. This area of impact is clearly very small (0.4%) compared to the total area of the deep Gulf of Mexico (1.2 million km^2), but a ten-fold increase in drilling activities would provide cause for environmental concern. In addition, if drilling targeted petroleum-seep communities, which contain some very long-lived species (e.g., vestimentiferan tubeworms which live over 100 yr; Fisher *et al.*, 1997), the highly specialized seep ecosystem could be substantially impacted.

It should be noted that current regulations in some countries require assessment of environmental impacts prior to drilling in deep water, and some high-profile, baseline environmental surveys are now underway in the deep sea (e.g., in the Gulf of Mexico and on the slope west of Scotland; Bett, 2001). As a consequence, some vulnerable deep-water habitats, such as the species-rich, deep-water coral (*Lophelia*) beds off Norway and Scotland, are likely to be protected from direct drilling impacts.

In the future, additional energy resources, in particular methane hydrates, are likely to be exploited in the deep sea. Methane hydrates are stable at ocean depths of 300 to at least 900 m, and constitute massive reservoirs of fossil fuel along many continental margins. Off the Chilean coast between 35^o and 45^{o}S, the methane gas potential (10^{13}-10^{14} m^3) exceeds by 3 orders of magnitude the annual consumption of gas in Chile (Esteban Morales, 2003). Worldwide estimates suggest that gas hydrates may contain twice the amount of carbon found in all other fossil fuels combined. Although this resource probably will not be exploited for at least another decade, development of hydrate extraction technology may dramatically increase the environmental impacts of deep-sea fossil-fuel exploitation, by releasing methane (a greenhouse gas) into the atmosphere, by disturbing specialized deep-sea communities associated with methane seeps, and by causing slope instability, potentially yielding turbidity flows, earthquakes and even tsunamis.

Manganese nodule mining. The deep seabed contains a number of potentially extractable minerals, including manganese nodules, which abound on the abyssal seafloor beneath the oligotrophic ocean. Manganese nodules are potential source of copper, nickel, and most notably, cobalt, which is used in jet engines and other high-technology applications. Although not expected to occur for another 10-20 years, manganese nodule mining could ultimately be the largest-scale human activity to directly impact the deep-sea floor. Twelve pioneer investor countries and consortia have conducted hundreds of prospecting cruises to investigate areas of high manganese nodule coverage in the Pacific and Indian Oceans, especially in the area between the Clarion and the Clipperton fracture zones, which extends over 6 million km^2 (Glasby, 2000) and may contain 78 million tonnes of cobalt, 340 million tonnes of nickel and 265 million tonnes of copper (Ghosh & Mukhopadhyay, 2000; Morgan, 2000). Seven contractors are now licensed by the International Seabed Authority to explore nodule resources and to test mining techniques in seven claim areas, each covering 75,000 km^2.

When mining begins, each mining operation could directly disrupt, through “strip mining” of nodules, 300-800 km^2 of seafloor per year (Oebius *et al.*, 2001, ISA personal communication), and disturb the seafloor biota over a very poorly constrained area perhaps 2-5 fold larger due to redeposition of suspended sediments (Smith, 1999; Thiel *et al.*, 2001). In any given year, nodule mining by one to two contractors could disrupt seafloor communities over areas of 600 to 8,000 km^2, and 15 years of such mining could conceivably impact 120,000 km$^{2.}$ of seafloor. The most obvious direct impact of nodule mining will be removal of the nodules themselves, which will require millions of years to regrow (Ghosh & Mukhopadhyay, 2000; McMurtry, 2001). Nodule mining would thus essentially permanently remove the only hard substrate present over much of the abyssal seafloor, yielding habitat loss and local extinction of the nodule fauna, which differs markedly from the fauna of surrounding sediments (Mullineaux, 1987; Bussau *et al.*, 1995).

Nodule-mining activities also will inevitably remove much of the top 5 cm of sediment, potentially redistributing this material into the water column (Oebius *et al.*, 2001; Thiel *et al.*, 2001). Many sediment-dwelling animals in the path of the collector will be killed immediately, and communities in the general mining vicinity will be buried under varying depths of sediment (Jumars, 1981; Smith, 1999; Oebius *et al.*, 2001; Sharma *et al.*, 2001; Thiel *et al.*, 2001). Because abyssal nodule habitats normally are physically very stable (possibly the most stable habitats on Earth) and are dominated by very small and/or fragile animals feeding on a thin veneer of organic matter near the sediment-water interface, some have postulated that the mechanical and burial disturbances resulting from commercial-scale nodule mining could be devastating (Jumars, 1981; Glover & Smith, 2003).

Because of the environmental risks posed by nodule mining, a number of *in situ* experiments have been conducted to evaluate the sensitivity and recovery times of these potentially very sensitive abyssal benthic communities to simulated mining disturbance. The experimental disturbances created were much lower in intensity and many orders of magnitude smaller in spatial scale than would result from commercial mining, but they provide some important insights into the sensitivity and minimum recovery times of abyssal nodule communities subjected to mining (reviewed in (Thiel *et al.*, 2001; Glover

& Smith, 2003; Thiel, 2003). It is clear that abyssal benthic communities will be substantially disturbed by even modest amounts (~ 1 cm) of sediment redeposition resulting from mining activities, and that full *sediment-community* recovery from major mining disturbance will take much longer than 7 years (and possibly even centuries). Unfortunately, these experiments do not allow us to predict the likelihood of species extinctions from nodule mining because we do not know the typical geographic ranges of species living within the nodule regions (i.e., we cannot say whether their ranges are large or small relative the potential spatial scales of mining disturbance). Preliminary data based on morphological analyses suggest that some common benthic species may range well beyond the scales of impact for individual mining operations (Glover *et al.*, 2002), but these results must be verified with molecular techniques due to the frequency of cryptic species in marine invertebrates (Knowlton, 2000). To fully predict and manage commercial mining impacts, substantially more information is required concerning species ranges, sensitivity to sediment burial, and the spatial-scale dependence of recolonisation in abyssal benthic communities.

Polymetallic sulfide mining. At present, there is commercial interest in polymetallic suphides in the deep sea as sources of gold, silver, zinc, lead, copper and/or cobalt (Glover & Smith, 2003). Polymetallic sulphides are generally associated with deep-sea spreading centers and occur in two forms: as metalliferous muds and as massive consolidated sulphides. Although metalliferous muds in the Red Sea sparked interest in the past (Degens & Ross, 1969; Thiel, 2003), current commercial interests focus on the massive sulphides around inactive hydrothermal sites at bathyal depths near New Guinea and New Zealand (Wiltshire, 2001).

Hydrothermal-vent communities are characterised by low diversity and highly localised biomass (Van Dover, 2000) when compared to both rocky and sedimentary substrata in other areas of the deep sea. In addition, vent communities appear to be highly resilient (i.e., they recover rapidly from disturbance). Although individual vent communities often are isolated and ephemeral, on a regional scale vents are continually forming and decaying, and the adaptations of the dominant vent species insures that new vents are colonized rapidly. Because of the ephemeral nature, high fragmentation, and high energy flux of vent habitats, the primary vent species often grow rapidly, reproduce early and disperse broadly (Lutz *et al.*, 1994; Tyler & Young, 1999). Nonetheless, as more vents are explored throughout the ocean basins, there is a growing realisation that the global vent ecosystem is characterized by distinct biogeographic regions (Van Dover *et al.*, 2002).

Mining of massive sulphides at active vents would be catastrophic to a local vent community, wiping out a large proportion of the fauna as well as modifying the stockwork (or "plumbing") that underpins the vent site. At mined vents, biodiversity might increase initially as scavengers come in to exploit the dead biomass created by mining. The impacts of vent mining would differ from those of nodule mining in the rate of faunal community recovery. At a vent site, even with the total destruction of the fauna and damage to the stockwork, new vents will form quickly. Colonization of vents sites proceeds rapidly (Tunnicliffe *et al.*, 1997, Shank *et al*., 1998) and within two years the vents would likely be colonised by primary species. Analogous vent colonisation and development has been analysed at the 9ºN site on the East Pacific Rise and this site shows a well-developed and evolving ecosystem 7 years after new vent formation (Shank *et al.*, 1998).

C. Waste Disposal

Solid Structures. Between 1914 and 1990, more than 10,000 ships have sunk to the seafloor as a result of warfare and accidents (Thiel, 2003). While the impacts of shipwrecks are poorly studied, it is clear that they may generate reducing habitats (Dando *et al.*, 1992), release petroleum hydrocarbons and other pollutants, and/or provide a habitat for hard substrate biota (Hall, 2001). With the exception of oil-laden tankers, most shipwrecks are likely to have a localized impact. However, the scale and duration of the effects of sunken oil tankers on deep seafloor communities clearly merits further study, especially in view of the recent sinking of the tanker *Prestige*, which lost 30,000 tonnes of oil and then settled at 3000 m off Portugal.

Recently, proposals to scuttle oil- and gas-storage structures in the deep sea have received considerable attention (Rice & Owen, 1998). Until both the negative and positive seafloor impacts of large structures, such as shipwrecks an oil platforms, are studied in more detail, it will be very difficult to compare rationally the relative merits of onshore versus deep-sea disposal of such structures (Dauterive, 2000; Tyler, 2003a).

Muntions and radioactive wastes. Over the last century, several million tonnes of conventional and chemical weapons have been dumped onto continental slopes surrounding the UK and other European countries (Thiel *et al.*, 1998). However, field surveys of major munitions dumpsites show little significant contamination of marine life, with many loose munitions and boxes colonized by sessile organisms (Thiel *et al.*, 1998). Existing munitions dumpsites are unlikely to have significant, widespread impacts on deep-sea ecosystems, but do pose a danger to deep-sea trawlers.

Low and intermediate-level radioactive wastes have also been dumped into the deep sea (Smith *et al.,*1988; Thiel *et al.*, 1998). From 1949 to the dumping moratorium imposed in 1983 by the London Dumping Convention, about 220,000 drums of low-level wastes were disposed in the northeast Atlantic by European countries, and over 75,000 drums were dumped by the USA in the North Atlantic and Pacific. Monitoring of the northeast Atlantic radioactive-waste dumpsites has revealed very little change in radionuclide levels, with a single anemone species exhibiting enhanced levels of ^{90}Sr and ^{137}Cs (Feldt *et al.*, 1985). At the USA dumpsites, measurable levels of radioactive contamination have been recorded in sessile suspension feeders (anemones), deposit feeders (holothurians and asteroids) and mobile predators (genadiers, bathypteroids and decapods) collected close to waste materials (Smith & Druffel, 1998). Significant transfer mechanisms of radionuclides into the deep-sea ecosystem include bioturbation into local sediments, bioaccumulation in deposit and suspension feeders, and uptake by mobile benthopelagic fish species (e.g., grenadier fishes) which may ultimately provide a pathway to humans through deep-sea fisheries.

Sewage sludge and dredge spoils. The disposal of sewage sludge at sea is permitted under the London Dumping Convention and offshore dumping of sewage sludge is now under serious consideration by a number of countries (Thiel *et al.*, 1998). The primary benthic impacts of sewage sludge disposal are likely to be animal burial, clogging of feeding apparatus, dilution of natural food resources for deposit feeders, increases in turbidity, toxicity from sludge components, reductions in bottom-water oxygen concentrations, and changes in community structure due to organic enrichment (Thiel *et al.*, 1998). The only studies of sewage sludge impacts in the deep sea have been conducted at the Deepwater Dumpsite 106 (DWD 106), located in 2500 m of water 106 miles offshore of New York (Van Dover *et al.*, 1992; Takizawa *et al.*, 1993; Bothner *et al.*, 1994).

Approximately 36 million tonnes of wet sewage sludge were dumped in surface waters at DWD 106 from March 1986 to July 1992. The distribution of sewage sludge at the seafloor was traced from silver concentrations, linear alkylbenzenes (wetting agents in detergents), coprostanol, and spores of *Clostridium perfringens*, which originate from mammalian feces (Bothner *et al.*, 1994; Thiel, 2003). During dumping, benthic macrofaunal abundance at the site increased significantly, although changes in total community structure were not dramatic (Grassle, 1991). Stable isotope ratios (delta ^{15}N) indicated that sea urchins and other megafauna assimilated sewage-sludge material beneath the dump site (Van Dover *et al.*, 1992). In addition, the levels of silver in sediments at DWD 106 were twenty-fold higher than in background areas, and contaminants appeared to penetrate to at least 5 cm into the sediment (Bothner *et al.*, 1994). Environmental concerns halted the dumping at DWD 106 in 1992; the subsequent recovery of stable-isotope signatures to pre-dumping levels in sea urchins (indicating a return to natural, phytoplankton-based foodwebs) has taken about 10 years (Van Dover *et al.*, 1992).

Sediments dredged from coastal waterways and harbors frequently contain high levels of contaminants such as hydrocarbons, heavy metals and synthetic organic substances (Thiel, 2003). The disposal of dredge spoils in the deep sea on continental slopes has occurred in a number of countries, and poses threats to seafloor ecosystems similar to those from sewage-sludge disposal. However, we know of no published studies of the impacts of dredge-spoil dumping on deep-sea communities, although at least one study is in progress in the northwest Pacific offshore of San Francisco (J Blake, personal communication). It is clear, nonetheless, that the often active nature of the slope biota and bioturbation rates on continental margins will cause many bioactive materials placed on the sediment surface to be rapidly incorporated into animal tissues and subducted into the sediment matrix (Levin *et al.*, 1999; Miller *et al.*, 2000; Smith *et al.*, 2000).

D. Climate Change

The earth has warmed by approximately 0.6°C during the past 100 years, and since 1976 the rate of warming has been greater than at any time during the last 1,000 years (IPCC, 2001; Walther *et al.*, 2002). A general warming trend is evident over large parts of the world ocean during the past 50 years (Levitus *et al.*, 2000). The Intergovernmental Panel on Climate Change (IPCC) suggests that by 2020, the climate will be a further 0.3 – 0.6°C warmer, and that average sea level will rise between 2.6 cm and 15.3 cm (IPCC, 2001).

The influences of changing climate on regional patterns of circulation, upwelling, and primary production in the surface ocean are very difficult to predict. However, it is now clear that many processes at the deep-sea floor are tightly linked to the surface ocean, allowing us to postulate how climate driven changes in epipelagic ecosystems may propagate to the deep-sea floor. In particular, deep-sea benthic processes appear to be tightly coupled to the quantity and quality of food material sinking from the euphotic zone, as well as to the variations in this sinking flux over regional space scales and seasonal-to-decadal time scales. For example, a number of studies indicate that benthic biomass and abundance, bioturbation rates, the depth of the sediment mixed layer, and organic-carbon burial in sediments co-vary with particular-organic-carbon flux to the deep-sea floor, and with primary production in overlying waters (Rowe, 1971; Emerson, 1985; Smith *et al.*, 1997; Glover *et al.*, 2001; Glover *et al.*, 2002; Smith & Rabouille, 2002; Smith & Demopoulos, 2003). In addition, various studies suggest that smaller, more rapidly reproducing biota such as microbes and foraminifera show rapid population responses to episodic (e.g., seasonal) food input to the deep-sea floor, whereas larger, more longer-lived taxa generally integrate seasonal changes and respond primarily to inter-annual to decadal changes in primary production and organic flux (Gooday, 2002). Thus, in very general terms, climatic changes resulting in increased near-surface productivity and deep organic-carbon flux may be expected to enhance benthic standing crop, bioturbation rates and depths, and carbon sequestration in deep-sea sediments.

Beyond this general prediction, the precise nature of ecosystem change (e.g., in biodiversity levels) will depend on many factors, such as the temporal scale over which changes occur, the quality of particles reaching the seafloor (e.g., fresh diatom aggregates versus heavily reworked marine snow), and the original composition of the benthic fauna. Different types of deep-sea communities (e.g., in the oligotrophic abyss versus within oxygen minimum zones) will respond in markedly different ways.

While it seems unlikely that drastic changes in deep-sea ecosystems will occur as a result of climate change in the next 20 yr, dramatic trends have been documented in deep seafloor communities that ultimately may be ascribed to climate change. For example, in the northeast Pacific at 3800 m, K. Smith and Kaufmann (1999) have found that the ratio between the sinking flux of particulate organic carbon and the consumption of such carbon by seafloor metabolism (i.e., a ratio indicating surplus or deficit of food settling as small particles to the seafloor), has progressively decreased over the period 1989 – 1996 from ~0.99 to ~ 0.30. This suggests that the proportion of food arriving as small particulate flux from surface waters fell by 2/3 over this 7-year period, while the metabolic demands of the sediment community remained roughly the same. Smith and Kaufmann (1999) postulate that this decline in deep POC flux between 1989 and 1996 could result from increasing sea-surface temperature in the eastern North Pacific (Roemmich, 1992; Smith & Kaufmann, 1999) which has led to a reduction in the supply of nutrients to the euphotic zone and a decline in the export of primary production to the seafloor (McGowan *et al.*, 1998).

In the North Atlantic abyss, a decade of trawl sampling has revealed a dramatic regional increase in the abundance of megafaunal holothurians, with one species (*Amperima rosea*) rising in density by up to three orders of magnitude (Billett *et al.*, 2001). This “regime-shift” in the structure of an abyssal megafaunal community may be driven by systematic changes in the structure of the phytoplankton assemblages and the quality of export production from the euphotic zone far above (Billett *et al.*, 2001; Wigham *et al.*, 2003). Similar cause-effect scenarios, in which altered phytoplankton community structure modifies deep-sea detritivore assemblages, may be repeated in many parts of the ocean as a consequence of anthropogenic climate change.

E. Impacts of Pollution

The deep sea is often considered one of the most pristine environments on earth, relatively unaffected by anthropogenic pollutants because of its distance from pollution sources, the slow rates of physical exchange between near-surface and deep water masses, and its vast diluent capacity. In fact, the deep sea and deep-sea organisms are an important sink, and often the ultimate one, for many of the most persistent and toxic of human pollutants. Halogenated hydrocarbons such as PCBs, DDT and related pesticide compounds, the trace metals mercury and possibly cadmium, and certain radionuclides show particularly high levels of bioaccumulation in the deep sea.

The pollutants of most concern in the deep sea are predominantly transported into the oceans through the atmosphere, due to their volatility or atmospheric emission, and thus achieve a global distribution. About 80% of PCB and 98% of DDT and related compounds enter the ocean through the atmosphere (Clark *et al.,* 1997). Approximately 25% of DDT enters the ocean within a year of production (Woodwell *et al*., 1971).

The substances of most concern are highly water insoluble, readily adsorbed onto particles and are generally lipophilic. Because of this behavior, they are rapidly incorporated into marine food webs and transported into the deep sea in zooplankton fecal pellets, moults and through the feeding and vertical migration of zooplankton and micronekton. The concentration of PCB in the faeces of the euphausiid *Meganyctiphanes norvegica* is 1.5 million times its concentration in the surrounding seawater (Clark *et al*., 1997), and methylmercury is concentrated by similar orders of magnitude in marine zooplankton (Mason *et al*., 1995). The residence time of organochlorides in near-surface waters is less than a year in the open ocean and as little as a few weeks in more productive waters, such as off Antarctica, before it is removed to deeper water (Tanabe and Tatsukawa, 1983). As a result, despite the great diluting capacity of the deep ocean, PCB concentrations are virtually uniform throughout the oceanic water column (Harvey *et al*., 1974; Tanabe and Tatsukawa, 1983).

Reasonably high levels of many pollutants are now widespread in deepwater organisms. PCB levels were consistently > 1 ppm and often > 10 ppm (on a lipid basis) across a range of dominant midwater planktivorous and small predatory fishes in the North Atlantic and Gulf of Mexico in the 1970's(Harvey *et al*., 1974b; Baird *et al*., 1975). Baird et al. concluded that "concentrations of pesticides and PCB's from midwater fishes were well within levels shown to be physiologically significant or even toxic to other fish species." Methyl mercury accumulates in species near the top of the food chain and in larger, older individuals. Mercury concentrations are several-fold higher in mesopelagic than in epipelagic planktivores (Monteiro *et al*., 1996), and concentrations in large, long-lived, deepwater fishes such as hake, orange roughy, sharks and grenadiers often approach or exceed maximum permissible levels for human consumption (0.5 ppm) (Barber *et al*., 1972; Cutshall *et al*., 1978; van den Broek and Tracey, 1981; Clark *et al*., 1997; Cronin *et al*., 1998). Significant behavioral impacts, such as inability to avoid predators, have been observed at mercury levels of 0.67 ppm (Kania and O'Hara, 1974).

Much about the cycling of pollutants in the deep ocean is poorly understood. Although global production of PCB was less than that of DDT compounds, the concentration of PCB in midwater organisms is generally several-fold higher (Harvey *et al*., 1974a). It is also poorly understood why very high levels of particular pollutants are found in certain organisms. For example, cadmium levels in black scabbardfish livers (6.98 ppm, wet weight basis) are 30-fold higher than in North Atlantic hake (0.22 ppm), another large midwater predator. (Both exceed permissible levels for human consumption of 0.2 ppm (Mormede and Davies, 2001).) Cadmium levels in the mesopelagic decapod shrimp *Systellaspid debilis* in the open Northeast Atlantic Ocean (~11-32 ppm) are > 10-fold higher than in other midwater crustaceans in the region (Leatherland *et al*., 1973; Ridout *et al*., 1985). Radionuclides such as plutonium from nuclear testing (and naturally occurring ^{210}Po) are also scavenged from near-surface waters, taken up into the food chain, and transported to deep water.

Public awareness has largely eliminated atmospheric nuclear testing and has dramatically reduced the productiom of PCBs, DDT, and mercury (although DDT continues to be used in developing countries); Nonetheless, deep-sea pollution remains a potential problem because the organochlorides are highly persistent (especially in long-lived deep-sea animals), and the concentration of mercury in the atmosphere and upper ocean remains triple that of 100 years ago (Mason *et al*., 1994). A number of pollutants continue to be at critical or near-critical levels in specific groups of organisms. However, pollutant impacts in the deep sea cannot be reasonably evaluated or controlled because the factors causing pollutant loading, and pollutant effects on behavior, physiology, genetics and reproduction, remain very poorly known for the deep-sea biota.

III. RESEARCH NEEDS

In order to predict and manage the effects of direct human activities and climate change, we must learn much more about the structure and functioning of deep-sea ecosystems. Some of the major knowledge gaps that severely hamper our predictive abilities of deep-sea environmental impacts are highlighted below.

A. Biogeography and habitat distributions

The biogeography of the deep-sea, and the distribution of its habitats, are very poorly known for at least three reasons. (1) *Deep-sea ecosystems are woefully under sampled.* For example, in the entire North Pacific Basin, we know of only 11 sites at abyssal depths (i.e., below 4000 m) from which macrofaunal species-level data have been collected (Smith & Demopoulos, 2003). For the South Pacific and Indian Oceans, we know of only *one* abyssal data set each (Ingole *et al.*, 2001; Thiel *et al.*, 2001). Substantially more sites have been sampled on continental slopes, at least in the northern hemisphere, but the slopes are much more variable then the abyss in habitat parameters, with benthic community structure varying substantially with water depth, substrate type, current regime, overlying productivity, seafloor efflux of reduced chemicals, and depletion of bottom-water oxygen. All of these parameters can change dramatically over distances of 10’s of kilometers on the continental slope (Levin & Gooday, 2003; Smith & Demopoulos, 2003; Tyler, 2003b), and, as a consequence, the distribution of habitats and species along the slope remain very poorly characterizied on regional to basin scales. (2) *Different sampling programs in the deep sea have often used different sets of taxonomic specialists to identify the animals collected.* Because many, if not most, species collected in the deep sea are new to science and have not been formally classified, it is often very difficult to relate the species list of one study to that of another, and thus to compare species and conduct biogeographic syntheses over basin scales. (3) *Most* *biodiversity and biogeographic studies in the deep sea have used traditional morphological methods for identifying species.* However, recently developed molecular methods (e.g., using DNA sequence data) suggest that morphological techniques typically underestimate the number of species and overestimate species ranges in marine habitats (Knowlton, 1993; van Soosten *et al.*, 1998; Creasey & Rogers, 1999; Knowlton, 2000).

Major programs designed to sample novel habitats and poorly studied regions of the deep sea (e.g., cold seeps, oxygen minimum zones, abyssal sediments in the South Pacific, hydrothermal vents and continental slopes in the Southern Ocean) clearly must be considered a research priority. Within such programs, it will be critical to apply modern molecular techniques to resolve phylogenetic relationships, levels of biodiversity, and species ranges. Perhaps most importantly, it will be essential to integrate the taxonomic collections of many different expeditions and countries to allow biogeographic syntheses on regional and global scales.

The biogeography of seamounts merits particular attention because seamount communities are potentially very diverse, highly variable across seamounts, heavily impacted by fisheries, and still very poorly understood. For example, as recently as 1987, existing data suggested that only 15% of the invertebrates and 12% of the fishes collected on seamounts were endemic (Wilson & Kaufmann 1987). However, while over 100 seamounts had been sampled, few had been studied comprehensively, and 72% of the species recorded came from only 5 seamounts. A mere 27 species had been collected from seamounts in the entire southwest Pacific. The first intensive South Pacific study (de Forges *et al.* 2000) reported more than 850 species of benthic macro- and megafauna from seamounts in the Tasman and Coral Seas: i.e., 42% more than had previously been reported from all seamounts worldwide since the *Challenger* expedition! Approximately 30% of these species were potential seamount endemics, and there was little overlap in species composition from one seamount chain to another (de Forges *et al.*, 2000). The authors speculated that seamount chains may be effectively isolated from each other due to topographic rectification of currents combined with the limited dispersal abilities of many deep-sea benthic organisms.

If general, the results of de Forges *et al.* (2000) have profound implications for the conservation of threatened seamount faunas. However, these results need to be tested in other oceanic regions and over broader spatial scales on the many distinct seamount chains that remain virtually unexplored (e.g., in the southwest Pacific, Indian and Southern Oceans). As for other deep-sea habitats, species composition and population genetic structure (e.g., gene flow within and between seamount chains) must be evaluated to allow reasonable conservation measures (such as the creation of marine protected areas) to be implemented.

B. Response of deep-sea ecosystems to human impacts

Numerous anthropogenic forcing factors will influence deep-sea ecosystems in the next 10-20 years (Table 2) and in some regions, especially continental slopes, several impacts may occur simultaneously (e.g., due to fishing, energy exploitation and waste disposal). This means that prediction of environmental impacts will be especially difficult because deep-sea community responses even to single stressors are very poorly known. Research priorities must include *experimental* evaluation of the response of deep-sea benthos to (1) oil drilling discharges, (2) burial, toxicant loading, and organic enrichment associated with dredge-spoil dumping, and (3) the disturbance, predator-removal, and bycatch dumping effects resulting from deepwater trawling. We emphasize that cleverly designed experimental studies will be necessary to resolve the response of deep-sea systems to varying levels of single and multiple stressors. It need also be recognized that community sensitivity and recovery processes may vary dramatically with habitat type across the deep sea (e.g, in oxygen minimum zones versus well oxygenated sediments), and will likely be inversely related to the physical stability and flux of organic carbon within the habitat. In particular, the very stable, food-poor habitats (e.g., the abyssal plains) are likely to be the most sensitive to, and slowest to recover from, from human impacts, but many slope habitats are also likely to be very susceptible to anthropogenic impacts compared to shallow-water systems.

It should be noted that research priorities should not be restricted to evaluation of specific environmental impacts, but must also be directed towards expanding our very limited knowledge of the general structure and dynamics of deep-sea ecosystems, especially those most likely to bear the brunt of human impacts. For a broad range of forcing factors and deep-sea habitats, there is a pressing need to identify (1) vulnerable species and ecological processes, (2) useful indicators of ecosystem health, and (3) the linkages between terrestrial, coastal and atmospheric systems that may channel anthropogenic stressors to the deep-ocean floor. In addition, the remarkable biodiversity of many deep-sea habitats merits special attention. Knowledge of the roles played by biodiversity in ecosystem functions such as stability, nutrient cycling, biomass production and carbon burial, is fundamental to conservation decisions (e.g., the locations and sizing of MPAs, gear regulation, and fishery and oil-lease closures).

C. Long time-series studies

To evaluate and manage environmental impacts in the deep sea, ecosystem variability resulting from natural processes must be distinguished from anthropogenic change. Because many human-induced impacts (e.g., global warming) may occur gradually, relevant natural variability can only be elucidated through time-series studies covering many years to decades. Time-series studies of ecosystem structure and dynamics are extremely rare in the deep sea, and have been restricted to only a few sites in the ocean (i.e., the northeast Atlantic and the northeast Pacific); nonetheless, these limited time-series data have documented surprising temporal variability in basic ecological parameters over multi-year time scales (Gage & Tyler, 1991; Smith & Kaufmann, 1999; Billett *et al.*, 2001; Ruhl & Smith, 2004). Clearly, the few existing deep-sea time-series must be continued, and new ones initiated, if we are to have a suitable ecological context in which to evaluate the deep-sea impacts of anthropogenic change in the coming decades. However, we must also recognize that it will not be possible to inventory and monitor all the organisms and regions of the deep sea. It would thus be highly desirable to develop key indicators of ecosystem health (e.g., particular taxa, assemblages, or biogeochemical processes) for specific types of deep-sea habitats, and to monitor these indicators at representative sites over long time periods to assess ecosystem change.

IV POLICY AND SOLUTIONS

Current and future anthropogenic forcing factors in the deep-sea ecosystem are many and varied (Table 2), targeting habitats ranging from slopes only a few miles off shore (e.g., dredge-spoil dumping off San Francisco) to the remote nodule-covered abyssal plains in the central Pacific (e.g., manganese-nodule mining). As a consequence, the policy recommendations and management approaches needed to control these varied impacts are far too complicated to consider in this short paper. We do, however, feel that the environmental protection is urgent in the deep sea because some deep-sea ecosystems (e.g., continental slopes and seamounts) have already been substantially modified by human activities, and many others will be profoundly altered in the near future.

Many policies and regulations to protect the deep sea must come from national governments because substantial portions of the deep seabed, including those habitats most likely to sustain immediate human impacts, lie within the 200-mile limit of Exclusive Economic Zones (EEZ’s). Nonetheless, most of the deep-ocean floor underlies the “high seas” (i.e.,waters outside of national jurisdictions) presenting unique policy challenges. The protection of seamount biota from the onslaught of deep-sea trawling in international waters provides an informative example of the challenges involved.

As a consequence of growing environmental awareness since 1999, Australia, New Zealand, Norway, the USA and other countries have declared deepwater Marine Protected Areas (MPAs) designed to safeguard deep-sea coral and seamount habitats. In December 2002, the UN General Assembly responded by passing a resolution recognizing the need to protect biodiversity (including seamount biota) on the high seas. How can this be accomplished?

Unfortunately, the United Nations Convention on the Law of the Sea (UNCLOS) specifically regulates mineral extraction on the high seas (the International Seabed Authority (ISA) set up by the United Nations is vested with that authority) but fails to address exploitation of biological resources. This was because in the 1970’s, when the UNCLOS was negotiated, the environmental implications of the global spread of deepwater fisheries was not yet appreciated. As a consequence, regulation of high-seas fishing and its environmental impacts has fallen into a legal limbo. However, several options are presently under consideration:

- Multinational, or UN, declaration of high seas MPAs to protect sensitive and unique environments
- A UN brokered global moratorium on demersal trawling on the high seas (or on seamount and deepwater coral habitats, in particular)
- Specifically vesting the ISA with the authority to regulate seabed biological resource exploitation (or at least demersal fishing) on the high seas.

These options are not mutually exclusive. The declaration of MPAs on the high seas is seen as lacking recognition under international law and appears unlikely to be instituted, even on a trial basis, within the next ten years. The advantage of a global moratorium on high seas trawling (proposed by Greenpeace) is its immediate blanket impact if adopted by the UN General Assembly. In the longer term, vesting an internationally recognized body, such as the ISA, with the authority to regulate the seabed impacts of biological resource exploitation could effectively close the gap within the international legal framework for the regulation of demersal fishing impacts on the high seas. Institution of a global and representative system of MPAs on the high seas might also be most effectively administered under the authority of the ISA.

V CONCLUSIONS

1) The deep-sea floor (i.e., the ocean bottom deeper than 200 m) covers most of the Earth’s solid surface. It harbors a broad range of habitats and remarkable levels of biodiversity, and yet remains largely unexplored.
2) Many deep-sea habitats and their resident biota are easily disturbed and slow to recover from anthropogenic disturbance due low levels of productivity, low biological rates, and delicate habitat structure.
3) A surprising number of deep-sea habitats, especially seamounts and continental slopes, are currently threatened by deleterious human activities such as bottom fishing, waste disposal, fossil-fuel extraction, and pollutant loading. The

environmental impacts of these activities, and the requisite conservation measures to minimize these impacts, remain extremely poorly understood and demand immediate attention.

4) Ecological processes at the deep-sea floor often are tightly coupled to the surface ocean, the atmospere, and the coastal zone. Climate change in the coming decades may thus alter deep-sea ecosystems on a global scale.
5) Because deep-sea research has lagged far behind that in other habitats, there is an urgent need to better understand key attributes of deep-sea ecosystems, including species distributions, community responses to anthropogenic disturbance, and natural temporal variability over years to decades. These needs can only be met by creative, well-funded research programs and are essential to predicting and managing human impacts in the deep ocean.
6) Anthropogenic impacts in the deep sea have substantially outstripped the predictive abilities of scientists, and this situation will worsen with accelerated human exploitation of the deep ocean. Enviromental protection in the deep sea, perhaps more than in any other habitat, will require application of the "precautionary principle," in which reasonable conservation measures are implemented prior to detailed scientific understanding of the ecosystem.

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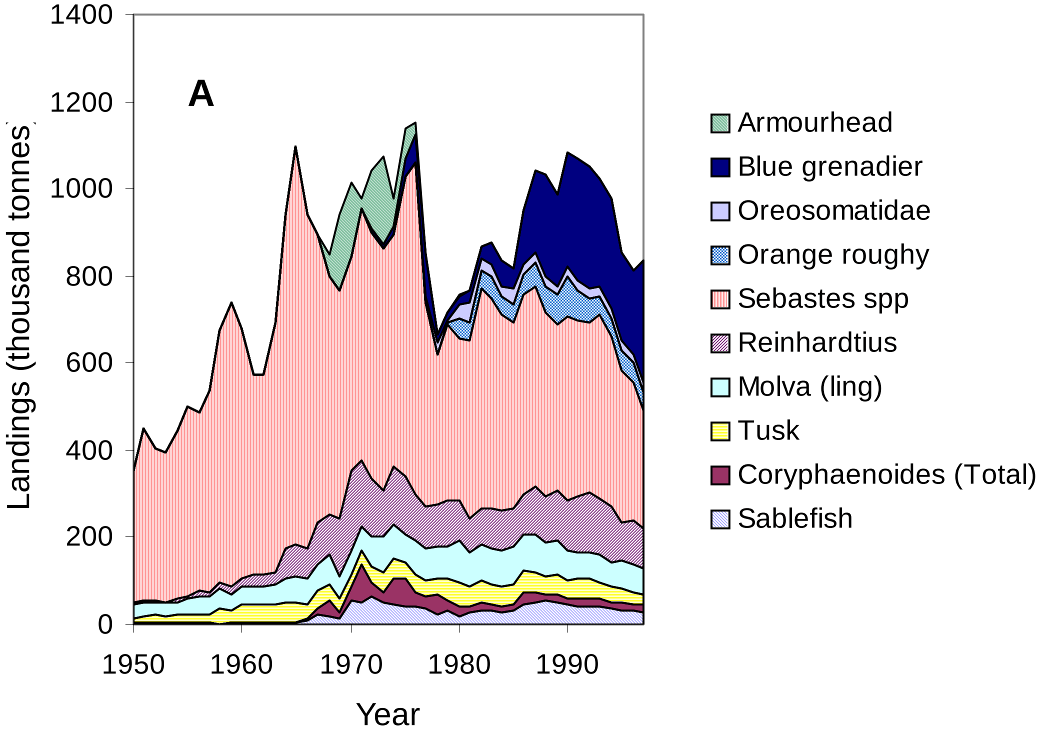
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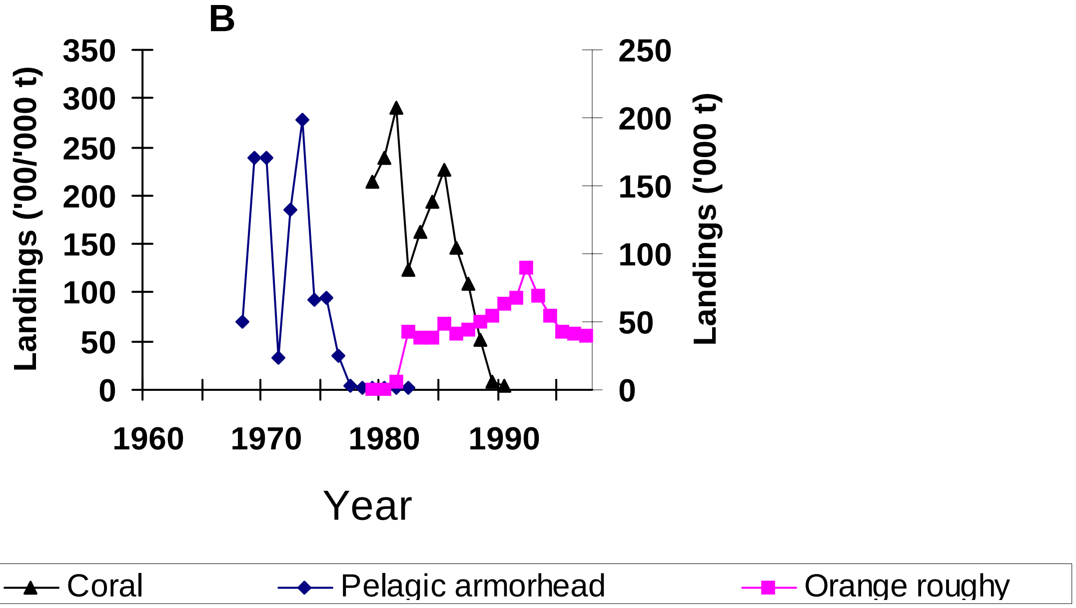
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50
0
Landings ('00/'000 t)
250
200
150
100
50
0
Landings ('000 t)
1960
1970
1980
1990
Year
Coral
Pelagic armorhead
Orange roughy

Figure 1

Figure Legends

Figure 1. A: Overall landings for deepwater fisheries, 1950-1999. B: Landings for the pelagic armorhead, orange roughy and *Corallium* sp. fisheries.

Table 1 A comparison of general ecological characteristics in the deep-sea relative to shallow-water (< 100 m water depth) ecosystems, and the consequences of these differences to ecosystem sensitivity to, and recovery rates from, anthropogenic impacts

Characteristic	Deep sea	Shallow water	Consequence to sensitivity to impacts	Consequence to recovery rates from impacts
Productivity	Usually low	High	Deep-sea more sensitive	Deep-sea slower
Growth, reproduction, recolonization rates	Usually low	High		Deep-sea slower
Biomass	Usually low	High	Deep-sea more sensitive	No effect
Physical energy	Usually low	High	Deep-sea more sensitive	Deep-sea slower
Size of habitat	Often large & continuous for 100-1000 km	Small, continuous only over 1-100 km	Deep-sea more robust (extinction less likely?)	No effect
Species diversity	Usually high on local scales	Moderate, variable	Deep-sea more sensitive	Deep-sea slower (more species to recover)
Species distributions	Broad?, but poorly known	Narrow, variable	Deep-sea less sensitive in some ways (exctinctions less likely?)	Deep-sea faster??

Table 2 Summary of anthropogenic forcing factors in the deep sea. Ranked according to the estimated severity level in 2025, within each category. Spatial scale of impact is indicated at the level of local (linear scale of 0-100km), regional (100-1000km) and basin (1000-10000 km). * In the deep sea, due to low biological and chemical rates, the time scale of deep-sea impact typically extends far beyond the the time scales of activity. For example, the impacts of a large shipwreck, or of deep seabed mining are expected to last >100 years.

Forcing Factor	Spatial Scale	Temporal Scale (estimates)*	Estimated Severity in 2025
Deep-water fisheries			
Direct effects of stock and biomass depletion	Regional – Basin	1950s onwards	High
Indirect effects (trawling damage, bycatch, whale removal)	Regional – Basin	1950s onwards	High
Energy and mineral extraction			
Polymetallic nodule mining	Basin	~2010 onwards	High
Deep-water oil and gas extraction	Basin	1990s onwards	Moderate
Polymetallic sulphide mining	Local	Present day onwards	Moderate
Manganese crust mining	Local	Unknown	Low
Methane hydrate extraction	Regional - Basin	Unknown	Low
Waste disposal			
Deep-water CO_2 sequestration	Local - Basin	~2015 onwards	High
Sewage and dredge spoil disposal	Local	~2005 onwards	Moderate
Oil and gas structures disposal	Local	Isolated incidents	Low
Radioactive waste disposal	Local	1950s – 1990s	Low
Munitions disposal	Local	1945 – 1976 (now banned)	Low
Climate change			
Changes in oceanic production	Global	~2050 onwards	Low
Other			
Research and bioprospecting at vents	Local	1970s onwards	Low
Shipwrecks	Local	Since shipping began, ongoing	Low
Underwater noise	Local	1960s onwards	Low