

VIEWPOINT

Environmental effects of marine fishing

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ABSTRACT

1. Some effects of fisheries on the associated biological systems are reviewed and management options and their inherent risks are considered.
2. In addition to the effects on target species, other sensitive groups impacted by fishing are considered including marine mammals, turtles, sea birds, elasmobranchs and some invertebrates with low reproductive rates.
3. Other impacts discussed include the destruction of benthic habitat, the provision of unnatural sources of food and the generation of debris.
4. Management options are considered including the designation of marine protected areas, risk aversion, and the burden of proof.
5. A balanced consideration of the risks and consequences of 'Type I' and 'Type II' errors is advocated.

INTRODUCTION

There is growing and widespread concern about the effects of overfishing on the populations of target species but little consideration of the more general effects of fishing on other ecosystem components. Thus this review focuses on some of the wider implications of the effects of fishing on marine communities and ecosystems. The objective is to consider the interaction between fisheries and the associated systems and to discuss some management options, their inherent risks, and their potential payoffs. Rather than offer a comprehensive review, the purpose is to emphasize the seriousness of this problem.

It is often presumed that exploiting a common heritage is more a right than a privilege. This presumption fails to recognize that the exploitation of marine resources may also adversely affect the general environment. The implications of such impacts on ecosystems are important and should determine what we consider to be 'acceptable' levels of use and corollary effects. It is essential to recognize that the risks include many factors in addition to the direct effects on the target species. Perhaps the most sensitive species impacted by fishing are species with low reproductive rates that may need decades to centuries to recover from serious deletions of their populations; these include species such as mammals, turtles, sea birds, elasmobranchs, and many benthic or deep-sea species. In addition, there are important impacts to the habitat itself. Many of these species and habitats have their own intrinsic value, are valued by other

resource users, and are of value to the public in general. Though inconspicuous, other components of these ecosystems are critical to the functioning of the rest of the system by serving as nurseries for settling larvae, important predators, competitors, etc. We argue that fisheries management must include the protection of these species and habitats as well as the target species.

BYCATCH, INCIDENTAL TAKE AND HABITAT DAMAGE

Introduction

Bycatch is perhaps the most serious general environmental impact of modern fisheries. Because the process is out of sight of the public and there are few objective studies, the data base is inadequate and attention to the problem has been limited. However, the issue is so important that there is increasing public concern and cooperation between the fishing industry and regulatory agencies (Schoning *et al.*, 1992; Alverson *et al.*, 1994), and we summarize growing literature (e.g. Hutchings, 1990; Andrew and Pepperell, 1992; Hall *et al.*, 1990; Hill and Wassenberg, 1990). A representative example is discussed by Pauly (1988) who reviewed the environmental consequences of trawling in Southeast Asia, where post-war fishing in the Philippines resulted in signs of overfishing by the 1950s. This was followed by considerable increase in trawling in the Gulf of Thailand. Pauly documents a classic rise and fall of the demersal fishery. One result was a reduction in mesh size; this resulted in a large trash fish industry from the bycatch. The 'trash fish' catch increased dramatically because high value shrimp subsidizes the harvest of fish at population levels much lower than would otherwise be economically feasible. That is, modern shrimp trawling operations led to a decrease in marketing of bycatch species and an increase in discards—thus high technology can add to the waste problem. And, consequently, this ensures a continued highly destructive bycatch.

Improved gear and technology continue to improve the effectiveness of the fisheries, and there have been coincident improvements designed to reduce bycatch. Yet these technical effects to reduce bycatch are juxtaposed with many technical innovations such as twin beam trawl, gill nets, paired trawling, etc., that also much increase the overall catch and inflict secondary damage. For those reasons effort data and data on the absolute amount of bycatch are extremely difficult to interpret.

There are important artifacts in many components of the bycatch literature. For example, some research surveys are undertaken in random patterns appropriate for stock analysis, but give a biased perspective of actual commercial and recreational fishing. Good fishermen do not harvest randomly; they are often brilliant in their ability to concentrate their effect on oceanographic systems such as fronts, Langmuir cells, and the benthic and water column effects of bottom heterogeneity ranging from pipelines and wrecks to natural reefs and seamounts. Such fishermen are extremely selective. Random surveys of research cruises may underestimate the actual environmental impact of the fishing activities. Another problem is that research surveys are often much more reduced in time and space than the actual fishing effort. This is done to obtain more replicates, reduce spatial variation of density estimates, and evaluate patchiness. However, these surveys also introduce an artifact by missing the high density patches of aggregated species likely to be taken by the larger scale commercial gill nets (Mangel, 1993). Most of the species particularly vulnerable to bycatch such as mammals, sea birds, turtles, and sharks occur in aggregations. ICES groups studying ecosystem effects of fishing activities have attempted to compensate for these problems, and these efforts continue (Anon., 1991b). It is considered that *all* efforts to evaluate bycatch and environmental effects of heavy fishing on natural systems are too late because most sensitive species have long been impacted, leaving no concept of natural relationships or patterns. There are in consequence few, if any, meaningful controls.

Bycatch problems are so pervasive that this review can only include summaries of a few examples of incidental take in particular habitats or of particularly endangered or threatened species. Specific habitats are considered separately.

Pelagic communities

General

Most pelagic bycatch occurs with net fisheries, however, even longline fisheries that are usually considered relatively free of bycatch, can actually result in important bycatch problems. For example, Freeberg (in Schoning *et al.*, 1992) outlines the history of the swordfish fishery of the northwestern Atlantic which incidentally took several times more sharks than swordfish. The catch of over a million large sharks was correlated with the estimated population of grey seals rising from 3000 to 45 000. This was consequently linked to high increases in gear destruction by seals and increased infections of cod with parasites for which seals are the primary host. In addition, such a population density probably induced stress on the seals and may in part have contributed to their large die-off in the late 1980s. Paterson (1990) reports that the Queensland shark netting programme from 1962 to 1988 resulted in an incidental take of 520 dolphins (a probable underestimate due to clerical errors), 576 dugongs, 3656 sea turtles, and 13 765 rays. All of these species have low reproductive rates which increases the effects of such mortality on the populations. Also, the highly endangered baiji, the Chinese river dolphin, is very seriously threatened by incidental take (Martin, 1990). Gillnets worldwide take a considerable toll of porpoise, and are directly implicated in the near extinction of several species (Jefferson and Curry, 1994).

The incidental catch of mammals, turtles and birds are of special concern because they are high profile species often protected by legislation, including in the USA, the Endangered Species Act and the Marine Mammal Protection Act. There are good biological reasons for concern about the conservation of species such as these, as well as sharks, rays, and the many deep-sea species that have life history characteristics of much delayed reproduction and low fecundity. Adult survivorship is extremely important to sustained populations of such species, and they are highly vulnerable to even moderately increased mortality. Northridge (1991) reviewed a study by Brander (1981) on a skate, *Raia batis*; the age of maturation of this skate is 11 years, after which it produces about 40 eggs per year. The size and shape of the fish make it vulnerable to all types of bottom fisheries from the moment it hatches, and it has apparently been essentially eliminated from the Irish Sea and is now probably extremely rare over its entire range. There are few data on bycatch of elasmobranchs because they are of marginal commercial importance or considered trash fish, but rays are known to have important community roles (Van Blaricom, 1982; Thrush *et al.*, 1991). Nonetheless, we may never know how species such as the Irish Sea skate interacted with its natural community.

Mammals

Net entanglements of the greatly depleted North Atlantic right whale are very serious; more than 50% of these rare whales are estimated to bear marks and scars indicating that they have encountered fishing gear (Kraus, 1990). Other high profile examples of incidental take of large cetaceans are found in the Arctic (Philow *et al.*, 1992) and in the Newfoundland coastal fishery where humpback, fin and minke whales are entangled in gill net and cod traps (Hofman, 1990). Almost 600 were taken between 1969–1986, but the annual take was apparently less than 1% of the population and appears not to be causing substantial declines in the populations (Hofman, 1990). Dolar (1994) surveyed a small area in the Philippines and reported massive takes of small cetaceans. A relatively small number of fishermen killed between 1900 and 2980 small cetaceans in the year she conducted her survey. When these are added to the several hundred more killed in a direct take (Dolar *et al.*, 1994) and this amount of mortality is projected over wider areas, it is apparent that small cetacean populations are gravely threatened.

The best known example of incidental take is that by the tuna purse seine fishery in the Pacific which is estimated to have taken over 6 million porpoise by 1987 with clear evidence that the porpoise populations were substantially reduced. Since then there have been many technological improvements, but the biological and political ramifications of this problem were staggering. Another example is that of Dall's

porpoise taken in the Japanese salmon drift-net fishery in the North Pacific. By 1987 approximately 5% and 20% of the Bering and western North Pacific Dall porpoise stocks had been taken in that single fishery (Hofman, 1990).

The high seas drift net fishery is another high profile example of incidental take that resulted in threatened UN action before it was closed in the North Pacific. This fishery covered several large geographical areas, had different target species, and different levels of bycatch (Alverson *et al.*, 1994). Estimates of incidental take were much compromised by the very low observer coverage in a situation in which most of the relevant species are strongly aggregated (Mangel, 1993). In such situations reliable estimates depend upon almost complete saturation of observers. The main conservation issue here was not the proportion or mass of bycatch compared with other fisheries, but was the saturation of the habitat with gill nets that efficiently killed vulnerable species. In this case the vulnerability is associated with rarity and lack of population resilience to adult mortality. Rare pelagic species such as large billfish, many species of marine mammals, turtles, and sharks are often highly mobile and efficient foragers likely to be attracted to nets with fish. Thus their incidental take will be out of proportion to the less mobile species. Species with life histories including slow growth rates and low reproductive rates such as mammals, sharks, turtles, and large sea birds are especially vulnerable because the incidental take eliminates the breeding population. For these reasons even very small bycatch relative to the target species can still be extremely important. The losses may have been very serious, and it is important that such bycatch be evaluated in terms of its impact on the populations of vulnerable species.

Perhaps the cetaceans most threatened by incidental take are the small coastal porpoises. The harbor porpoise, for example, are taken by gill nets (Polachek, 1989), as are Hector's dolphins in New Zealand; this mortality is much higher than the population can sustain (Dawson and Slooten, 1993). Read and Gaskin (1988) report catch rates of approximately 0.1 harbor porpoise/km of net/day; probably an underestimate because the populations are poorly known, as are the actual amount of gill nets set along the coast. Still, the bycatch effects are very serious (Anon., 1993a). Northridge *et al.* (1991), report heavy take of harbor porpoise in Europe (for example, 500–1000 annually in Danish waters), and Palka (1994) and Anon. (1993a) document sufficient take to list the species. Burmeister's porpoises and the spectacled porpoises are impacted in South America by gill nets, as are the Chinese finless porpoises (Jefferson and Curry, 1994). The vaquita or Gulf of California harbor porpoise has been so reduced by coastal gill nets that they may be the next marine mammal to become extinct (Barlow, 1986; Silber, 1988; Brownell *et al.*, 1989; Vidal, 1990). To prevent this, Mexico recently declared the entire northern Gulf of California a reserve. Hopefully other countries will follow this example of protecting species threatened by indirect fishing mortality.

Other examples of threatened marine mammals include the highly endangered Mediterranean and Hawaiian monk seals (the Caribbean monk seal is extinct). Although the Hawaiian monk seal has declined along with other higher predators in the presence of changing oceanographic conditions and attendant reductions of appropriate food (Polovina *et al.*, 1994), the seal's decline also coincides with fishing pressure in the Northwest Hawaiian Islands. As these fisheries have developed there have been substantial decreases in pup and juvenile survivorship associated with evidence of interactions with nets and long lines. A very modest survey programme in 1992 recorded 14 monk seals entangled with fishing gear. There are also several cases of seals observed with hooks embedded in their mouth or skin and also bearing injuries thought to have been caused by fishermen attempting to recover gear (Nitta and Henderson, 1993; Anon., 1991d). California sea otters, as well as thousands of sea birds and non-target fish (including at least 15 species of sharks and rays), were killed by gill and trammel nets, and for several years the estimated population of sea otters fell (Bishop, 1985). In 1990 the state of California prohibited use of gill nets shallower than 30 fathoms throughout most of the sea otter range, and the otter population began another recovery. Both the North Pacific fur seal and the northern sea lion populations appear to be affected by interactions with fisheries which include incidental take in high seas nets, trawling near hauling areas, and perhaps resource competition for a variety of fatty fishes (Alverson, 1992). Much the same is true of

Hooker's sea lions in New Zealand and Sub-Antarctic waters. Finally, the river dolphins are some of the world's most threatened aquatic mammals and most of their decline seems to result from little noticed bycatch of various types of fishing. The baiji (*Lipotes vexillifer*) has received some attention (Perrin and Brownell, 1989; Kaiya and Xingduan, 1991; Ellis *et al.*, in press), and its situation is probably typical of all others as the habitat is fragmented by dams and development. The small, increasingly isolated populations are being eliminated by fishermen, especially by long-liners.

Turtles

Sea turtles and other marine organisms that feed on or around target fishery resources frequently get caught as bycatch, are inadvertently killed or injured through contact with fishing gears (Henwood and Stuntz, 1987; Chan *et al.*, 1988; Duronslet *et al.*, 1990; Pointer *et al.*, 1990), or are forced to modify their behaviour as avoidance or in response to fishing-related stress (e.g. noise). Loggerhead turtles, for example, show a strong food preference for shrimp and crabs and are thus often found in close association with these organisms. The same is true for Kemp's ridley turtles, similarly opportunist feeders sometimes found in parts of the Gulf of Mexico where shrimp are densely aggregated. Both Kemp's ridley and loggerhead turtles suffer high rates of fishing-induced mortality through incidental capture in shrimp trawls. This fishing-induced mortality is very damaging for two reasons: (1) these species are already threatened with extinction—especially the Kemp's ridley exhibiting a worldwide population size of possibly only 1000 breeding adults, thought to be dangerously close to the threshold for minimum viable population size; and (2) the population dynamics of these species are such that adults and subadults (most often captured in fisheries operation) are several hundred times more valuable than juveniles in terms of population replacement potential (Crouse *et al.*, 1987). Fisheries-related mortality may be the single biggest factor preventing recovery of sea turtle species (Anon., 1990a).

Sea turtle-fishery interactions are not restricted to shrimp fishery-related incidental catch, though this fishery impact may be among the most studied (Anon., 1990a). Sea turtles often drown from entanglement in net and line gears as well. Longline sets are especially attractive to pelagic turtle adults—although data on turtle by-catch in longline fisheries are extremely limited. However, a '10 June 1993 National Marine Fisheries Service Section 7 Consultation' issued a 'Biological Opinion and Incidental Take Statement' that concluded that incidental take rates of up to several tens of thousands of sea turtles are possible in the Hawaiian-based longline fishery alone. More recently the use of glow sticks by swordfish long liners has attracted leatherback turtles resulting in heavy incidental take of these large and endangered animals. Even pot and trap fisheries cause mortality to turtles: leatherback turtles often mistake marker buoys for jellyfish and become entangled in buoy lines. Gill nets cause indiscriminate mortality of all species of sea turtles. Such gear can be a significant cause of mortality and impaired population recovery when such fisheries are undertaken in breeding areas or migration corridors.

There is growing concern that nursery habitats for most sea turtle species are also vulnerable to environmental degradation caused by fishing. Carr (1987) suggested that continental shelf or open ocean convergence zones and Langmuir cells may be a critical habitat for hatchling and juvenile turtles. These areas suffer from chronic pollution from ships and land-based debris, and are target areas for certain fisheries. Reliance has been placed on stranding data, but most turtles simply sink and only a very small but unknown percentage get to a beach where they can be counted by stranding networks. These data represent such an underestimate as to be almost useless as a measure of mortality. The fact that the turtle nesting sites are subject to vandalism and intense predation as well as poaching of the females accentuates the severity of impacts caused by fisheries-related mortality. These factors combine to emphasize that the mortality at sea is extremely serious for populations of all sea turtles worldwide.

Seabirds

In some areas gill nets have killed a large proportion of local diving birds; for example, the 1991 ICES (Anon., 1991b) report discusses the loss of some 900 razorbills and divers in St. Ives Bay in 8 days. Most

nets in the North Sea are too short to be forced to register, but several thousand kilometres of net are imported annually (Anon., 1991b lists many areas with massive bird kills). Data regarding total kills of any species in gill nets are rare because of lack of observers; and there are no records on total amounts of gill nets set. Data on seabird entanglements are particularly rare because many incidentally caught species, though having important community roles, are not afforded special protection or attention as threatened species.

Tuna long line fishermen in the southern hemisphere are thought to take many tens of thousands of albatrosses and have been implicated in a massive annual kill of at least 44 000 wandering albatross (Croxall, 1990). The actual annual take may be twice as high (Brothers, 1991). Several authors have discussed the relationship between fisheries and seabirds in the California Current and in Alaskan waters (Ainley and Hunt, 1991, Ainley and Sanger, 1979; DeGrange *et al.*, 1993; Salzman, 1989; Springer, 1992; Takekawa *et al.*, 1990), and there is a great deal of evidence of heavy incidental catch of seabirds as well as inferential evidence of resource competition between seabirds and fisheries. Ainley *et al.* (1994) review the other sources of the population declines of seabirds, especially breeding site problems. They concluded that the relationships between fisheries and declining seabird populations were qualitative but extremely persuasive. They argued further that such fisheries may cause many complicated population interactions between seabirds and mammals. As always, this entire issue needs much more research, but we already know that the populations of most long-lived birds such as wandering albatross depend upon high adult survivorship, and they simply cannot sustain such massive mortality. These problems simply cannot wait for more research if these populations are to persist.

Benthic communities

There are many types of trawls, dredges, and traps that sit on or are dragged over the sea floor. Bottom fishing gear is not selective and bycatch is a serious problem. The effects on the sea bottom include impacts such as scraping and ploughing the bottom to substratum depths of 30 cm as well as causing resuspension of sediment and destruction of many bottom organisms (Riemann and Hoffmann, 1991; Jones, 1992). Bergman and Hup (1992) report that a beam trawl can remove bites at least 6 cm into the bottom, and the boards of otter trawls get as deep as 15 cm. They report long lists of benthic species destroyed, and that most good areas are trawled over many times a year. Their study area was fished at least three times per year, and their experimental three-fold trawling reduced echinoderms, polychaetes and molluscs by 10–65%. Northridge (1991) reviews some of the European literature that suggests considerable incidental mortality and loss of target species such as scallops and other molluscs, and that bycatch can be extensive. The ICES report (Anon., 1991b) summarizes many cases of extensive benthic bycatch in the North Sea. In most cases the benthic mortality is extremely variable, but often very high. Bergman *et al.* (1990) found target species formed only one-fifth to a third of the total catch. Bergman and Hup also report almost complete mortality of the long-lived bivalve *Arctica islandica*, and they review studies showing that the damaged specimens are consumed by cod and other predators. Even a 25% mortality is extremely serious for long-lived species that recruit episodically and live in areas exposed to trawling several times a year.

Scallop dredging can be expected to impact benthic animals (Thrush *et al.*, 1993), but because it is often done in deeper water, it is little studied. However, Caddy (1973), Chapman *et al.* (1977) Dupouy (1982), Holme (1983), Bullimore (1985) and Rees and Eleftheriou (1989) all report very substantial mortality of target species, bycatch, and especially the almost complete loss of sessile species occurring on rocks and cobbles. Scallop fishing grounds in relatively deep areas with a high diversity of encrusting species on boulders and rocks are likely to be particularly prone to dredge disturbance. A recent review of seabed trawling (Jones, 1992) covers many facets of the problem including indirect effects such as the turbidity killing *Platinopecten* and *Pecten* scallop larvae, the elimination of slow growing deep-water coral *Lophelia*, and the destruction of bryozoan beds which serve as fish nurseries. Eleftheriou and Robertson (1992) carried out an experimental scallop dredging programme in Scotland. In this shallow (10 m) depth there

were no apparent effects on topography or sediments nor on motile pericardid crustacea or shallow burrowing clams. However, sessile forms such as polychaetes and *Echinocardium*, a spatangid echinoderm, were substantially reduced. More important, the large fauna, including the mollusc *Ensis*, the asteroid *Asterias*, and the large and important cancer crabs were heavily damaged and killed, as were large concentrations of the sand eel, *Ammodytes*. Comparison of box cores before and after trawling indicates extensive damage to the infauna, especially *Echinocardium*. Also, tube building polychaetes may be losing as much as 50% of their populations, and in some areas they are important nurseries for larval recruitment. The fact that the incidental take of *Arctica* and *Echinocardium* has continued so long suggests that it might consist of juveniles spawned by an adult population existing at sediment depths below the dredges. If this is the case, the continued elimination of recruitment could eventually eliminate the populations.

Hydraulic dredging damages almost all the infauna (MacKenzie, 1982; Poiner and Kennedy, 1984; Van Der Veer *et al.*, 1985). One vessel fishing with a hydraulic dredge retrieved over 4000 tons of stones and gravel for a yield of 3 tons of scallop meat (Anon., 1991b). The substrata is sieved on board and most of the contained animals are killed by the heat. Essentially all the benthic animals exposed to such techniques die and most sessile species are largely eliminated by hydraulic dredging. It seems likely that many of these species serve as larval nurseries, and the mechanical bivalve harvest damages the habitat (Peterson *et al.*, 1987).

Trawling differs from dredging in that gear is dragged on or near the bottom to recover benthic or near-benthic species in the water column or on the soft bottom. Its effects are also extensive and potentially severely damaging to the ecosystem. De Groot (1984) and Jones (1992) report concern about trawling effects on benthic communities as early as 1376. Despite the long history there has not been much study of this important problem and there is little documentation of potential destruction because the fishing effects must be distinguished from the often large natural variation in time and space. Because all areas that can be fished have been impacted for so long, it has proven extremely difficult to find otherwise reasonably similar control areas not impacted by fishing. Time-series data long enough to demonstrate changes in relation to fishing are rare. However, Reise (1982) and Riesen and Reise (1982) discuss many changes since the 1920s in the Wadden Sea which include the loss of oyster bed and polychaete reefs, and Holme (1983) reported extensive trawling induced degradation of benthic communities in the English Channel.

Otter trawling often produces a great deal of bycatch, especially crabs and scallops. The Dutch BEON group (Anon., 1990c; 1991c; 1992) showed mortality rates of 10–30% for starfish, 10–50% for many molluscs, 40–60% for crabs and over 90% mortality of the clam *Arctica islandica*. Note that these are mortalities for a single trawling capture. Extrapolation of these data to annual trawling intensity suggests staggering levels of benthic mortality. Then, if we factor such mortality rates with the age of maturation, this level of mortality may virtually eliminate many species from these habitats. Many other common benthic genera such as *Tubularia*, *Lagis*, *Ensis* and *Solen* can be heavily impacted (de Groot, 1984; Dyer *et al.*, 1983). Hamon *et al.* (1991) report high mortality of crabs and scallops. Langton and Robinson (1990) demonstrated important effects on scallop beds including shifts in macrofauna density and sediment types over large areas.

The ICES report (Anon., 1991b) discussed attempts to estimate the areas affected by towed gears. They summarize types of gear, penetration and areas fished per 100 h; the total area of the study region is calculated to be 346 811 km² yr⁻¹ with major impact and 667 572 km² with contact in 1989. The percentage coverage of total North Sea trawl fishery areas affected by beam trawls and otter trawls varies from 0.3% to 321.0% with the overall average being 34%. Analysis shows that some areas are very heavily fished, but overall less than 60% of the bottom was trawled because fishermen concentrate effort on good areas and avoid areas where gear may be lost. Scallop dredging is more serious; British boats increased efforts between 1974 and 1989 from 132 to 1600 km² of English Channel, and the Bay of St. Brieuc, France (total area is 800 km²) experiences impacts on 160 and 5600 km² by scallop dredges and otter trawlers respectively. In the latter case the bay is trawled over as many as 7 times yr⁻¹ (Hamon *et al.*, 1991). Rauck (cited in

Bergman and Hup, 1992) calculated that several North Sea habitats are trawled 3–5 times yr^{-1} , and Krost *et al.* (1990) estimated 25% of the Kiel Bight and 70% of the Dutch North Sea had visible trawl tracks (Anon., 1992b); Welleman (1989) calculated that some areas were trawled 0.5 to 7 times yr^{-1} . Churchill (1989) estimated that areas of Long Island and Narragansett Bay were trawled some three times yr^{-1} .

The physical destruction and alteration of the habitat from trawling has not received much attention. Northridge (1991) reviewed bottom fisheries that have destroyed *Zostera* beds and saltmarsh vegetation, horse mussel beds and their extensive associated invertebrate community, as well as many types of molluscs, crustacea and echinoderm dominated communities. He reviewed a Scottish example in which all the epifauna, a population of long-lived bivalve *Ensis* and a calcereous algal bed of maerl were destroyed by eight passes of a dredge. Epifaunal species are especially vulnerable, and Northridge reports trawlers destroying sea pens and beds of the reef building polychaete *Sabellaria spinulosa*, the oyster *Ostrea edulis*, and sea grass *Zostera marina*. The same patterns have been observed in the *Posidonia* beds in the Mediterranean. Holme's (1983) English Channel work discussed the loss of hydroid and bryozoan habitats. In New Zealand, Bradstock and Gordon (1983) also reported the loss of large beds of bryozoans as a result of trawling. In each of the above cases the habitats that were destroyed by trawling probably represent very important nursery areas for many species, often including some of the target species of fisheries.

In addition to direct impacts, there are many indirect impacts caused by dredging or trawling resulting from increased turbidity likely to reduce or eliminate the remaining sea grass habitats. In most cases these are important habitats that become dominated by small deposit feeding polychaetes. Such shifts have serious implications because deposit feeding communities may resist recovery of suspension feeding species. Epifauna often play key roles in influencing the structure and stability of benthic communities. They can modify benthic boundary flow characteristics which further influences sediment characteristics and the deposition of larvae. The heterogeneity these organisms create provides a refuge for a variety of species, especially juveniles, from predators. Analysis of the bycatch of a fin-fishery on the Australian Northwest shelf (Sainsbury, 1988) showed a decrease in the number and variety of epifauna, particularly sponges, collected over time, with shift in the fishery from high to low value species. Such disturbances have been underway for decades and possibly centuries; one can only speculate what such habitats might have been naturally.

Alteration of sediment type is another important effect of bottom fishing. This is the predominant means of eliminating *Zostera* and contributes to many of the community changes discussed above. Langton and Robinson (1990) found significant declines (70% for scallops and 20–30% for burrowing anemones and fan worms) resulting from a scallop fishing induced shift in sediment from organic-silty sand to sandy gravel with quantities of shell hash. Much of Caddy's (1973; 1990) work also documented smothering of suspension feeders by benthic fishing. Churchill (1989); and Churchill *et al.* (1994) have shown important sediment changes resulting from trawling in deep shelf edge habitats, where fine sediments are not naturally transported by currents. Even in shallow habitats, where sediments and associated organisms can be suspended by storms, organisms removed by fishing may play important roles in stabilizing the sea bed, making the impact of these natural disturbance events much more pronounced. Other direct effects on sediments include modifications to microbial activity (Meyer *et al.*, 1981), resuspension of contaminants, and increases in benthic/pelagic nutrient flux (Krost *et al.*, 1990).

Over large scales, gradients in hydrodynamics and food supply select for species with particular functional and life-history patterns characterizing 'functional groups' (Rhoads and Young, 1970; Rhoads, 1974). Such groups include (1) suspension feeders such as clams, (2) tube building species which can affect substratum stability and recruitment of many species, and (3) detritus or deposit feeding groups. These groups have very different but substantial effects on the substratum. In almost every case in which a long-lived suspension feeding group (bivalves are the usual target) is intensively disturbed there is likely to be a

shift to a habitat dominated by detritus feeders. Once the detritus feeding group becomes established, they can resist recovery of suspension feeders by consuming and smothering the potential recruits. Diversity of functional groups can be enhanced by small-scale disturbances (Probert, 1984; Hall *et al.*, 1994), but larger scale disturbances will have markedly different effects. Most fishing impacts result in the development of short-lived deposit feeding associations. Thus habitat disturbance by fishing gear that removes surface dwelling organisms, modifies sediment topography, and occurs over large scales will result in reduced heterogeneity in benthic communities. Over time repeated intense disturbance will select for species with appropriate facultative responses, and communities are likely to become dominated by juvenile stages, mobile species, and rapid colonists. These features have far reaching implications for marine ecosystems and are likely to predispose them to destabilizing influences.

Deep-sea

Thanks to new technological developments and decreasing costs of technologies that allow ever-greater access, deep-sea habitats are subject to increasing amounts of fishing. These communities are characterized by life-history adaptations such as slow growth, extreme longevity, delayed age of maturation, and low natural adult mortality. Also they often are characterized by fragile structures that have important community roles (Levin *et al.*, 1991). Such adaptations are characteristic of systems with low productivity and turnover; they are extremely vulnerable to human intervention such as fishing (Messieh *et al.*, 1991; Thiel and Schriever, 1990), and there is a considerable risk attendant to any disturbance in this habitat. Again, documentation is usually lacking, but Jones (1992) reports frequent reduction in invertebrate fauna in trawls working deep water habitats; in some cases this is associated with declines in juvenile fish following removal of bryozoans. In another case, changes in the composition of pair-trawl fisheries followed losses of sponges, alcyonarians and gorgonians. Anecdotal stories of the New Zealand orange roughie taken in spawning aggregations over deep-sea pinnacles at about 1000 m depth report that when the fishery began the trawls brought up a great deal of benthic life; but almost all such incidental take has ceased (Jones, 1992). Little is known about epifauna on seamounts, but it seems clear that they recruit and grow very slowly (Genin *et al.*, 1986). Intact seamount communities may provide a critical area for aggregation, courtship and/or mating, and spawning of pelagic animals (Genin *et al.*, 1986). While such reports are anecdotal, they probably reflect important habitat changes with consequences beyond the immediate damage.

Lissner *et al.* (1991) reviewed the substantial though mostly grey literature of the US Outer Continental Shelf (OCS) Program covering the shelf from 60–300 m in depth, and integrated the findings into the relevant theoretical literature that emphasizes the differences in recolonization and recovery between type 1 and 2 disturbances (Connell and Keough, 1985). Type 1 disturbances result in death of some residents leaving a patch at least in part bounded by survivors; type 2 disturbances are larger, resulting in patches isolated from existing assemblages. Lissner *et al.* (1991) considered many types of disturbances and point out that the recovery in smaller type 1 disturbances will often be from margins, emphasizing the local community. Such succession results from local factors including (a) vegetative growth, (b) asexual budding, (c) settlement from fast growing opportunistic species which will often disappear, (d) short lived larvae from adjacent areas, (e) long lived larvae from distant slow growing species, and (f) immigration of motile adults. On the other hand, recovery from type 2 disturbances will be much slower and will emphasize (a) opportunistic fast growing species, (b) long-lived larvae from slow growing species, (c) asexual reproduction from motile species such as *Metridium*, and (d) immigration. Type 2 recovery will involve a stronger stochastic element and might produce a rather different yet very persistent patch. Furthermore, the OCS literature suggests that sediment encroachment in type 2 patches is relatively common and long lasting. These larger disturbances reflect those expected to result from deep water fisheries and may fundamentally alter the community. Given the extent of the extremely heavy trawl fisheries, much of the world's shelf communities may have already been altered.

Coral reefs

The effects of fishing on coral reefs vary from altering the size structure of target fish to cascading effects on other reef fish species composition, biomass, and density (Sebens, 1994; Hughes, 1994). Russ and Alcala (1989) document many direct and indirect effects of intense fishing on abundances, species richness, and distribution of other fishes as well as other benthic invertebrate species. Almost all components of the reef system are heavily impacted by the fishing. Indeed, Ormond *et al.*, (1990) offer several lines of support for the hypothesis that the reduction of several predacious fish species from intense fishing has contributed significantly to the destructive *Acanthaster* outbreaks along the Great Barrier Reef. In Kenya sea urchins are very destructive to coral reefs. It appears that urchin populations are naturally reduced by fish, and that with the locally extremely heavy fishing, the urchin populations expand and damage the reefs. The urchin densities could be 100 times the natural levels when the fish are so reduced (McClanahan and Muthiga, 1988; 1989; McClanahan and Shafir, 1990).

Fishing on coral reefs has become extremely damaging to the reefs themselves. A recent international workshop of coral reef experts ranked overfishing as the most important hazard (Roberts, 1993), especially when dynamite is used to blast the reefs and stun fish. This involves the loss of the reef structure that offers important protection from storm waves as well as protection from predators, breeding and nursery areas, etc. Nonselective poisons also have been used to kill fishes; all have widespread community consequences (Saila *et al.*, 1993). Weber (1993) reports that a single corporation in the Philippines has been responsible for 40 muro-ami ships that collectively destroy as much as a $1 \text{ km}^2 \text{ d}^{-1}$ of reef. More recently, high densities of traps with small mesh size are effectively removing most of the fishes. There is a growing literature describing these problems summarized by Russ (1991), Wilkinson (1992), and Hughes (1994). Coral communities have already shown the effects of extreme destabilization with cascading effects of the grazing by *Diadema*, an active sea urchin (Hay, 1984), and the predation on the corals themselves by *Acanthaster planci*, a voracious asteroid that can kill almost all the hard corals in a given area (Moran, 1986) and *Drupella*, a corallivorous gastropod also capable of devastating corals (Turner, 1994). The mechanisms causing these population oscillations are not known and no doubt vary; however, some, especially the release of *Diadema*, may reflect heavy fin fishing. Clearly it is likely that the perturbations will be further aggravated by co-occurring human disturbances (Hatcher *et al.*, 1989). Eutrophication, sedimentation, ocean warming, and other anthropogenic impacts can cause cumulative stress that, when coupled with even low-level sustained fishing pressure, may cause severe functional damage to reef systems (Agardy, 1993).

SECONDARY EFFECTS OF DISCARDS

The Northridge reviews and various ICES reports document that in some fisheries there can be very high proportions of discard from target species processed at sea. This material is returned to the sea where crabs, fish, mammals and birds often aggregate to consume it. This has certainly affected the natural behaviour of the scavenging species such as the fulmars in the late 1950s (Anon., 1991b). Because only some species utilize this resource, it has a selective effect on the communities and may put other species at a competitive disadvantage. Jones (1992) reviews examples in which trophic relations are changed by fishing. For example, the heavy shelled bivalve *Arctica islandica* formed a substantial part of cod and flatfish diets in Kiel Bay only after trawling began, because the fish were feeding on clams crushed and uprooted by otter trawls (Arntz and Weber, cited in Jones 1992). Medcof and Caddy (1971) and Caddy (1973) found intense feeding on exposed and damaged animals along trawl tracks. Jones (1992) considers that conflicting observations result from the use of different gear, but more modern gear has become increasingly heavy and destructive.

In most fisheries the vast majority of discarded organic material is from bycatch. Large amounts of biomass is discarded; this affects marine ecosystems in the same way as does organic pollution from other

human activities and often has many secondary effects. Northridge (1991) reviewed several studies that document benthic effects of discarded bycatch. Extrapolation of the few good studies suggested that the total discard biomass approximates and often far exceeded that of the landings. For example, Jones (1992) reviews Australian data collected by Wassenberg and Hill (1990) showing that prawn trawlers discard 3000 tons of material, mostly crustaceans and echinoderms, for each 500 tons of prawns; most of this discard sinks to the seafloor potentially to cause oxygen depletion problems. One study in Norway (Oug *et al.*, 1991) reported far-reaching effects on the benthic community that lasted at least 3 years. In general there are several potential effects of dumping organic material ranging from the aggregation of predator species to local anoxia. These effects are likely to be most pronounced in areas with low current flow or in situations where discarded material is deposited on sensitive communities and habitats. Jones (1992) reviews some of the literature showing that small-scale vertical oxygen gradients can be critical; for example Arntz and Rumohr (1982) demonstrated that an elevation of only 30 cm above the substratum allowed survival of normal fauna killed on the bottom. Finally, Jones reviewed situations in which discarding bycatch changes the behaviour of organisms such as lobsters as well as an Australian situation in which decomposing material apparently caused a disease that eliminated a scallop fishery. Benthic organisms have a clear relationship with the sediment with which they are associated, consequently one can expect cascading and possibly long-term effects from dumping large amounts of organic material. One potential result is to change the 'grain' or patchiness of the benthic habitat to select for highly motile predators such as fish or crabs that are quick to locate and consume isolated patches rather than other predators such as relatively slow moving asteroids. Such issues have never been studied, but there are likely to be important benthic community consequences of replacing asteroids with artificially inflated densities of scavenging fish and crabs.

Apart from the material landed on fishing boats, heavy fishing gear modifies food availability in many other ways (Berghahn, 1990). Dredges and trawls expose and damage animals which normally live buried in sediments, thus making them more susceptible to predation. Caddy (1973) reports large numbers of fish and crabs attracted to feed on animals exposed by dredging. Indirect mortality from fishing gear is often significant even for target species. One study of a scallop fishery in Australia (McLoughin *et al.*, 1991) demonstrated that only 11.6% of scallops in the tow path of a dredge were caught, the rest of the stock was wasted through direct and indirect mortality resulting from dredging. Quite apart from the wastage of resources, little is known about whether these combined mortality rates are too great for the target species to sustain. Van Beek *et al.* (1990) report low survivorship of plaice and sole discarded in the north sea, and Kaiser and Spencer (1994) report that gurnards and whiting respond to beam trawl damage to urchins, scallops and clams. They note that beam trawling creates food resources for opportunistic species, and that this could alter long-term community structure.

Hydraulic dredging is even more damaging; Medcof and Caddy (1971) found tracks 20 cm deep which were full of broken shellfish and other invertebrates which considerably alter the natural foraging patterns. These effects are likely to be most pronounced in areas with low current flow or in situations where discarded material is deposited on sensitive communities and habitats.

INDIRECT EFFECTS OF THE REDUCTION OF TARGET SPECIES

The removal of prey by fisheries may result in the loss of resources for other predator populations such as was seen in the collapse of the Peruvian guano birds following the loss of much of the anchovy stocks (Northridge, 1991). In general, such situations are more complicated than this example because fishing might change the schooling behaviour of the prey such that dense schools are simply scattered, with important consequences to their predators (Brock and Riffenburgh, 1960; Murphy, 1980). For example, tightly schooled sand lance and krill aggregations are important to fin and blue whales that are unable to capture efficiently dispersed prey (Brodie, *et al.*, 1978). Similarly, balls of jellyfish may be important to

leatherback turtles. The foraging behaviour of many species of small cetaceans and sea birds supports the hypothesis that aggregated prey are important; for this reason fishing induced dispersal of the aggregates is likely to be a problem for many types of predators.

Because predators are near the top of the food chain, their removal is an obvious concern. Perhaps because pelagic food webs are relatively unstructured, there are few examples of the cascading effects of predator removals that one finds in some benthic systems. However, because overfishing has been so extreme in many regions, the effects of missing species must be evaluated (Northridge, 1991). The removal of southern ocean baleen whales represents one example which resulted in a release and reallocation of krill to seals and birds and probably squid. Unfortunately, even this example is poorly studied. Hofman (1990) speculates that over the years some 700 000 tons of whales are thought to have been removed from the Gulf of Maine, as well as millions of tons of large fish. It seems likely that such selective harvest has impacted the remaining food webs. Other examples include the functional removal of sea otters with cascading effects through kelp forests (Simenstad *et al.*, 1978). Clear indication of cascading effects of fishing are discussed by Pauly (1988) who reviews an example of an indirect effect of a destructive demersal fishery in the Gulf of Thailand. Pauly documents the collapse of the target species, the virtual disappearance of rays and sawfish both as a result of bycatch and the loss of their food base, and the subsequent increase of snappers and squid. There seems to be a pattern in tropical demersal fisheries in which the reduction of the target stock is followed by an increase in squid, probably because the demersal eggs and very young of the squid are released from predation. Indeed, the squid stock sometimes continues to rise in spite of fishing pressure.

Several other examples in which the removal of marine predators by fisheries appears to have had an impact on the trophic structure of the community include (1) the exploitation of herring and mackerel that result in smaller fish and in some cases reduced abundance of whales, (2) exploitation of Bering Sea pollock affecting mammals and birds, and (3) Peruvian anchovies affecting birds (Parsons, 1992). Lowry *et al.*, (1989) discuss many pollock-mammal relationships; Ainley *et al.* (1994) also discuss at length the possible relationships between fisheries, seabirds and mammals. A more recent Alaska Sea Grant workshop report (Anon., 1993b) evaluated Bering Sea and northern Gulf of Alaska data and concluded that the declines of many marine mammal and bird populations were associated with lack of food. Interestingly, during this period some of the finfish were increasing. It is difficult to separate natural changes from those resulting from resource competition with fishing, especially in situations in which the fishing effects also include a considerable incidental take of mammals and birds as well as bycatch of alternate prey otherwise available to the mammals and birds. That is, most such papers evaluate the effects of fishing by consideration of landed catch, but this may be but a subset of the real problem. Certainly it is clear that food web changes and cascading ecological effects of heavy fishing are likely to impact the ecosystem. Simply because such impacts were exerted in the past and old or recent field data are lacking does not imply that the effects are any less important. This issue must not be ignored.

Most benthic systems are responsive to the removal of predators, but because most areas of the world's continental shelves have been subjected to extensive fishing for many centuries, it is very difficult to understand what natural situations may have prevailed before the heavy fishing. For example, within two or three hundred years after the Vikings colonized Iceland, they had virtually eliminated large cod from the coastal system (Thomas H. McGovern, personal communication). Witman and Sebens (1992) have demonstrated differences between coastal and distant reefs in the Gulf of Maine that they attribute to the intensive New England coastal fisheries which have removed enough large predators to release a different suite of benthic predators such as echinoderms and crabs. A similar relationship with echinoderms and crabs has been discussed by Langton and Watling (1990). Both cases suggest widespread and important community effects. Massive intertidal community changes have been demonstrated to result from the removal of predators by fishermen in Chile (Castilla and Duran, 1985; Moreno *et al.*, 1984, 1986). The functional elimination of southern king crabs may have released sea urchin grazing in South American kelp

habitats in a manner similar to that of the reduction of sea otters in the North Pacific (Dayton, 1985). Because no controls are available, these suppositions are indirect, but often they are supported by strong inferential arguments and corroborative evidence. Their effects can not be discounted simply because overfishing has eliminated controls. Indeed, Aronson (1989, 1990) argued that such overfishing has virtually eliminated the evolutionarily new teleost predators, resulting in a rebirth of the Mesozoic-like system dominated by echinoderms and crustacea.

A common rationalization for the collapse of a population is that it collapsed for natural reasons, would have collapsed without the fishery, and is independent of the fishery. The evaluation of natural vs. fishery related mortality is often considered in the fishery literature, but irrespective of the natural mortality, the fishery based mortality is almost always a significant component of failed fisheries (see Francis, 1986). Obviously the natural mortality and natural recruitment failures must be evaluated, but they are rarely independent. The case of the Alaska king crab represents an interesting example of the argument that natural changes in the marine environment are responsible for the dramatic reductions of an exploited population rather than the fishery *per se*. The crab population experienced an apparently natural failure of larval recruitment in the mid to late 1970s at a time when the stocks were at an all time high (D.L. Alverson, personal communication). Coincidentally, a large crab fishing fleet was developed, and began a heavy fishery that included a high bycatch of young crabs that suffer mortality when thrown back (D.L. Alverson, personal communication). This combined with a bycatch of crabs by other fisheries resulted in a collapse by the mid-1980s. Thus several phenomena including fishing contributed to the near collapse of the population; subsequently it has been very slow to recover. Large king crabs have no important natural predators; in general adult survivorship is an important adaptation to situations with unpredictable recruitment (Stearns, 1992). Proper management of such species, including most large animals, must ensure that an adequate year class is available before the reproductive adults are harvested. Because the fishery largely eliminated the adult life history component responsible for persistence in the face of natural recruitment failure, it was probably chiefly responsible for the virtual loss of these large predators. While not specifically studied, such predators are likely to have been important functional members of the benthic community. This almost certainly has substantial ecosystem ramifications because we know that king crabs once travelled in large groups, ate almost everything in their path, and must have exerted significant effects on the habitats. As for many potential impacts of fisheries, logistic problems prevent experimental documentation of the community roles of such predators, but this alone does not mitigate their significance.

Similarly, one unknown factor for coral reefs is the effect of the removal of the large predators such as groupers and basses. These predators reached weights of hundreds of kilograms and were apparently common. They and most other large piscivores are now largely eliminated from reef systems, and this may have resulted in an undocumented increase in herbivorous fishes. The removal of carnivorous and herbivorous fish in coral habitats has many indirect effects including impacts on sea urchins, plant growth patterns, diversity, and patterns of abundance and distribution of plants, corals, sponges and tunicates (Hay, 1984; Levitan, 1992; Wilkinson, 1992; Hughes, 1994). Selective removal of algal grazers on the reef, that are naturally widely dispersed and thus 'rare' per unit area, also results in widespread community effects such as unchecked algal overgrowth acts to suffocate coral polyps and associated organisms (Hatcher *et al.*, 1989). Beyond the coral reefs, one might also consider the natural impact of very high densities of marine turtles and manatees and dugongs (in the IndoPacific) on the reef systems. Early Spanish records mention extremely high densities of these large animals in the Caribbean (Randall, 1965; Jeremy Jackson, personal communication), and they must have exerted many substantial ecological effects. But they have been gone so long that we now consider the distribution and condition of turtle grass beds to be natural. By the late 1950s the only grazing of herbivores on turtle grass in the Caribbean occurred in the presence of reef refuges for the small fish and urchin herbivores (Randall, 1965).

As top predators, target species tend to be those that can be expected to have had ecosystem roles. Yet, as always, there are cascading effects of large scale oceanographic shifts and other natural features (Barry and

Dayton, 1991). Effects of El Niños are also conspicuous (Arntz *et al.*, 1988; Tarazona *et al.*, 1988). These natural perturbations are important and can have a devastating effect on some oceanic populations, especially through synergistic interactions with heavy fishing.

GENERATION OF DEBRIS AND THE ENVIRONMENTAL CONSEQUENCES

The drift net fishery in the north Pacific set 30–40 000 km of nets a day; Eisenbud (1985) estimated a daily lost rate that totalled over 20% of these lost each year. Lost gear and other fishery debris are widely suspected to have important and long-lasting effects on marine populations (Shomura and Yoshida, 1985; Smolowitz, 1978; Laist, 1987, 1994). The ICES report (Anon., 1991b) reviews efforts by Canada and Norway to document such impacts. The Canadians responded to complaints by fishermen on Georges Bank and made 236 grapnel tows; of these some 8% retrieved some 341 actively fishing ghost nets (Brothers, 1992). They found that the length of time prey persisted in nets averaged between 2 and 5 days. Given such a high turnover rate, this level of ghost fishing must have an important impact on bottom species. The Norwegians found that nets may continue to fish for many years; nets lost in 1983 were still fishing in 1990. As in the Canadian study, many nets had fresh fish. An unpublished ROV survey off California reported about 1% of the bottom littered with fishing debris, much of it actively fishing (A. Lessner, personal communication). Off New England 9 lost gillnets were found in 0.4 km², continued to catch fish and crabs and did not tangle up over 3 years (Carr and Cooper, 1987). In 1990 and 1991 the Bristol Bay king crab fishery lost 31 600 pots with a minimum loss of over 200 000 pounds of crabs, not to mention bycatch (Kruse and Kimker, 1993), and at least 11% of Dungeness crab traps were lost in one year in British Columbia (Breen, 1987). Crabs can have very important ecosystem effects (Thrush, 1986).

The Canadians evaluated some other effects of fishing related debris and found 260 incidents on 1070 trips on the Scotian Shelf. Half of these were porbeagle sharks fouled with strapping bands, most of which came from bait containers. Similarly, in 38 days one Faroe Island long-liner caught 26 porbeagle sharks fouled with packing bands (Anon., 1991b). Slip and Burton (1991), reviewed observations of the same problem for southern fur seals in Antarctica. Sea birds are also impacted: about 3% of all live gannets at Helgoland were entangled and 29% of all dead ones had been entangled (Schrey and Vauk, 1987). Furthermore, almost all seabird nests that have been checked near Helgoland and Newfoundland have remains of fishing gear, much of which is a danger to chicks (Montavecchi, 1991). A Japanese survey reported 217 ghost fishing nets on the surface along a 220 000 nm track (Anon., 1991a); this is a conservative estimate because most are below the surface. Finally even as far south as the Sub-Antarctic Heard Island there was 1 net/120 m of beach (Slip and Burton, 1991). Croxall (1990) and Croxall *et al.*, (1990) also report heavy incidental take and entanglement of birds and fur seals in Antarctic habitats. This illustrates the intensity of fishing in otherwise pristine isolated habitats.

COASTAL HABITATS AND TERRESTRIAL–MARINE INTERACTIONS

Several habitats are extremely vulnerable to anthropogenic disturbances that often include fishing. Intertidal and shallow subtidal communities are diverse and resilient to small scale perturbations; they are, however, vulnerable to large scale disturbances because they cover very limited areas, are near population centres, and in most areas are intensely fished. Reef habitats are often isolated by soft bottom habitats; they represent small islands that are heavily exploited and disturbed by people. These disturbances include the virtual strip mining of some of the most important species such as bivalves, gastropods, asteroids, sea urchins, and even sea weeds by recreational collecting in addition to fishing. Where these disturbances have been evaluated they have proven very serious (Moreno *et al.*, 1984; 1986; Ortega, 1987). The effect is that the small scale resilience depends upon dispersal from undisturbed habitats, and in many areas they have

ceased to exist. Worldwide these habitats are massively altered by human activities, and many heavily impacted coastal or nearshore areas represent sites where ecological processes are concentrated.

Subtidal rocky habitats are characterized by encrusting communities resistant to predation and invasion, but because most of the species have very poor dispersal (Olson, 1985; Jackson and Strathmann, 1981; Witman and Sebens, 1992), they too are extremely vulnerable to larger scale disturbances including trawling as well as sedimentation and pollution. Fanelli *et al.* (1994) report an appalling example of desertification along most of the Apulian coast in southern Italy resulting from the destructive fishing for mussels that has degraded much of the substratum. Rothschild *et al.* (1994) report massive losses of Chesapeake Bay oyster habitat resulted from destructive practices including dredges and hydraulic powered tongs that destroy the rock substrata.

Coastal wetlands and bays are probably the world's most endangered habitats and are especially vulnerable to human disturbance and habitat destruction because they are often near population centres and locations where inputs are not rapidly dispersed. As they depend on diffusion of propagules from the ever-dwindling number of other bays and wetlands, the remaining habitats are extremely vulnerable to even low levels of sport fishing. An additional, often unrecognized problem is that in many larger bays and harbours, the native populations are often almost completely replaced by exotic species. Some of these have been intentionally introduced, but many may have come via ballast water in ships (Carlton and Geller, 1993). Thus in these areas the interactions between the activities of fishermen and other resource users is very evident.

Coastal shelf communities are characterized by populations with very broad biogeographical ranges. Many also have good dispersal potential and excellent recoverability from small scale disturbances. But they too are massively altered by coastal fisheries, especially trawl fisheries that destroy so much habitat that recovery is difficult. The motile epibenthic or demersal shelf species are little studied by ecologists, but fishing related disturbances probably have affected most such species on the continental shelves.

MANAGEMENT CONFLICTS AND SOLUTIONS

Introduction

Management involves two overriding objectives: (1) striving for the optimal sustainable use of resources and maintenance of natural values of the long term, including the preservation of genetic diversity; and, (2) preserving the integrity of the ecosystem, both its structure and function. The preservation of these attributes includes managing the many human effects on marine environments. In addition to the normal ramifications of harvest, it also includes preservation of cultural, spiritual, or philosophical and aesthetic values. Management for the integrity of the ecosystem is extremely difficult because most systems have been fundamentally altered by the removal of top predators and by habitat destruction, and in many coastal areas the cumulative effects of civilization including pollution, habitat and nursery destruction, sedimentation, etc. may completely alter the natural ecosystems.

Ludwig *et al.* (1993) briefly review the history of the exploitation of wild resources and conclude that the ideal of sustainable use has not yet been achieved, and that the only real constant in the history of resource exploitation is that 'resources are inevitably overexploited, often to the point of collapse or extinction'. This essay has stimulated the invitation of many thoughtful commentaries compiled by Levin (1993); apparently not invited were government fishery scientists, some of whom responded independently (Rosenberg *et al.*, 1993). These responses amply support the contention that ecologists rarely enjoy a consensus. In this case the consensus is unlikely because that is a long history of failed efforts to obtain solid agreement about fishery management. Such failures are often the result of managers ignoring scientific advice. Caddy and Gulland (1983) and Caddy and Sharp (1986) describe a ratchet effect whereby harvests stabilize during normal periods and fishermen quickly gear up for good periods so as to maximize the return during unusual

conditions. This results in the creation of jobs and acquisition of expensive equipment and large debts during the good period. The ratchet effect and subsequent overcapitalization takes place because these exploiters cannot gear down for normal periods, much less below average periods, and it has proven politically difficult to reduce the catch. Thus when a good period ratchets up the exploitation there is intense pressure for subsidization during poor periods. Nevertheless, in principle sustainability is possible and desirable (Rosenberg *et al.*, 1993).

Risk aversion

The ICES (Anon., 1991b) report considers that a common fishing policy's Total Allowable Catch (TAC) approach does not work very well because it does not regulate discards. This can be especially damaging in purse-seine caught pelagic species such as mackerel or young haddock. The Advisory Committee on Fishery Management (ACFM) of ICES has recently described their charge (their italics) '*to provide the advice necessary to maintain viable fisheries within sustainable ecosystems*' and explicitly endorses the precautionary approach of erring on the safe side. Clearly this is a major step in the right direction. The ICES report also notes that 'perhaps whatever major changes did result from fishing occurred many decades ago'. And later: 'this is not arguing that a new equilibrium has necessarily been established. Rather the present levels of perturbation constitute the normal condition for the duration of our data series'. And later: 'Even if the full extent of the ecosystem effects of fishing remains unclear, . . . the exploitation of the living resources . . . undoubtedly affects the structure and functioning of the ecosystem and must therefore be viewed against other management objectives.' Clearly they share the concern about long-term ecosystem changes.

The principal challenge to the management of any wild resource is to incorporate the uncertainties and to allow maximization of the catch in such a way that the exploited stock is neither wasted nor put at risk. In most cases the exploited fish stocks experience recruitment uncertainties as well as ecosystem changes that alter growth productivity. In many cases not only do the target species in the USA appear to be in decline, but most fisheries also impact much of the rest of their ecosystems through bycatch and habitat disruption. Many of these other component populations also are in serious decline.

Economic considerations are always important and they usually have a strong impact on management. In this situation the role of science is to inform management of the condition of the stocks and ecosystems, the nature of any uncertainties, and the risks of different management options given the uncertainties. This allows management to make informed decisions, and helps ensure their accountability for the decisions. It is important to consider that the risks include many factors in addition to the well being of the target species. Some of the more important factors include long-lived species, especially mammals, turtles, sea birds, and elasmobranchs that have low reproduction rates and may need decades to recover. In addition, there are other large scale considerations such as ecosystem stability and productivity, animal rights issues, etc.

The use of statistical power analysis in management has been advocated by Peterman (1990), Peterman and M'Gonigle (1992) and Taylor and Gerrodette (1993). The thrust of this argument is that there is usually a null hypothesis of 'no effect' of a perturbation examined by a statistical test which can reject the hypothesis and conclude that there is an effect of the fishery on some other component of the ecosystem. If we conclude there is an effect when there is actually no effect, we have made a *Type I error*. Scientists try to reduce the frequency of such errors to less than 0.05. If we do not reject the hypothesis of 'no effect' when one exists, we have made a *Type II error*. Most fisheries and other environmental management programmes focus on reducing the probability of making a *Type I error* and virtually ignore the probability of making the *Type II error* of failure to recognize a real impact. Such analyses require that representative samples have been collected on the appropriate scales—these in themselves are complex and important issues.

Scientific advice to policy makers should be equally explicit about both types of errors and their probable consequences. The *Type I error* of modifying fishing practices when in fact it has no important impact is

loss of revenue; the *Type II error* of not making modifications is some highly *probable* effect on the target species and various components of the ecosystem. It is important to offer the policy maker probable consequences of both types of error *and* the recovery time to compensate the error. The *Type I error* results in loss of revenue; while there may be a lag to gear up, the recovery time usually will be very fast. The temporal consequences of the *Type I error* are limited. The consequences of the *Type II error* of continuing to fish at the same level when it is having an impact will often include the virtual loss of a resource and serious ecosystem effects for which recovery may take decades. It is important to emphasize the differences between recruitment and growth productivity and explain that some species such as turtles, mammals, marine birds and most populations with slow growing individuals will take a very long time to recover should management make a *Type II error*. Policy makers understand very clearly the financial implications of making *Type I errors*, but the ecological importance of the *Type II error* can only be predicted from inferential data. There are likely to be economic considerations here too, particularly because fisheries are integrated components of natural systems. It should be made very clear that in most cases these are very strong inferences. Probably the best means of reducing *Type II error* can also be achieved by a better understanding of ecosystem functioning. Certainly the importance of balancing risks and collecting representative data at the right scale are issues relevant to all resource management/conservation problems, not only fisheries.

Peterman and M'Gonigle (1992) discuss the fact that the scientific bias towards eliminating *Type I errors* is compounded with the legal tradition placing the burden of proof on the regulator. This has the effect that even in the face of strong inference of damage, the degradation must be extremely severe before action can be taken. This legal bias means that most research will be focused on the elimination of *Type I error* because there are so many uncertainties the exploiters can use to prevent effective management. Peterman and M'Gonigle also discuss 'surprise effects' as one of the problems resulting from such bias. This refers to the situation in which impacts (or management decisions) are assumed safe and only later found to be damaging. They point out that these anthropogenic surprise effects are escalating in number and severity.

It is remarkable to observe the strength of the conviction of groups exploiting public resources that their exploitation is an unalienable right, even if they are destroying other resources that take much longer to recover than the target resources. Considering the fact that most of these resources belong to society as a whole and are being managed by representatives of society, all the resources, consumptive and non-consumptive, should be managed to protect the most vulnerable component. Because this often means a reduced profit, such logical management is fought at both political and private levels. Finally, considering that there is a significant profit potential, there is no logical reason why society (either via the government, conservation groups, or even private citizens) is responsible for the burden of proof. The assessment should be based on the very best scientific information available, and it would seem reasonable to ask those profiting from the exploitation of the public's resources to bear the responsibility of evaluating the possible impacts and the risks. Certainly other industries are subject to a variety of environmental impact assessments and regulations.

Another important type of problem making environmental protection exceedingly difficult, if not impossible, under the present social and legal climate is that of cumulative, often low level disturbances. Cumulative effects from pollution, habitat fragmentation, and cascading ecological responses are extremely difficult to study. Because solid data are lacking, various other cumulative effects are often blamed for the environmental damage more likely resulting from overfishing. For these and many other reasons, effective environmental protection at the system level is extremely difficult, but important.

Marine protected areas and reserves

Recent work indicates that the conditions required for fishing to cause evolutionary change are met in most fisheries (Policansky, 1993). For example, growth overfishing may result in strong selection of smaller sized

individuals, and reduce genetic variability as discrete stocks are successively fished down. The ICES report (Anon., 1991b) mentions that orange roughie and salmon may have lost genetic diversity by heavy harvesting, and there are a number of studies showing that heavy fishing is altering the life-history characteristics of North Sea stocks. Such conclusions about genetic alteration in response to heavy fishing are not surprising, but noise in natural systems makes analysis difficult. Perhaps the most rigorous analyses have been done with salmon, reviewed by Ricker (1981) and Smith *et al.* (1991). Additional evidence regarding important fishing induced life history changes in salmon has been published by Gjerde and Gjedrem (1984), Healey and Heard (1984), Hankin (1985), Hankin and Healey (1986) and Hankin *et al.* (1993). Empirical substantiation of the intuitively and theoretically clear potential of such genetic changes is difficult, but the salmon work is sufficiently convincing that this issue must be addressed in most responsible fishery management programmes.

Marine fishery reserves offer one management tool (Bohnsack, 1992; Roberts and Polunin, 1993). An interesting perspective (Anon., 1990b) for reef fish management applicable to many non-pelagic stocks identified several major problems: (1) loss of potential recruitment because of insufficient spawning stock; (2) increased probability of recruitment failure due to environmental uncertainty and shorter generation times; (3) loss of genetic diversity within species resulting in less desirable stock; (4) massive over-fishing of many species; (5) declines in abundance and average sizes of fish; (6) loss of biotic genetic diversity; (7) potential disruptive reef fish community instability and permanent alterations; and (8) faster selection against desirable traits due to shorter generations. The authors suggest establishing completely protected Marine Fishery Reserves (MFRs) that do not even allow catch and release fishing. The idea is to protect older and larger fishes that are important to the maintenance of original genetic stock. This approach protects critical spawning stock biomass, intra-specific genetic diversity, population age structure, recruitment supply and ecosystem balance. Fishery reserves, if sufficiently large, numerous, and appropriately placed provide insurance against management and recruitment failures, simplify enforcement, and have equitable impact among users. MFR sites with natural species equilibrium will allow study of age, growth and natural mortality, elucidation of important natural interactions in the ecosystem, and provide a basis for educational benefits. Bohnsack *et al.* (Anon., 1990b) consider minimal habitat areas and recommend that 20% of the continental shelf should be a reserve. Such restricted reserves also act as signposts for long-term changes and help separate natural and anthropogenic changes (Davis, 1989). The number, locations and sizes of the reserves must be calculated on a case-by-case basis, but they should include all the habitat types, and the smallest boundary should be no less than 20 miles (32 km); compared with the exploited areas, this is relatively small.

Marine protected areas that allow certain types of harvest represent another management tool (Agardy, 1994). Such MPAs are established for various reasons, often to achieve multiple management goals simultaneously. These are described as larger areas designed to serve as starting points for exploring and delimiting functional linkages in coastal systems. Perhaps more importantly they are designed to test multiple use adaptive management procedures that can be realized by societies that need to both use and protect their coastal habitats (Salm and Clark, 1984; Gubbay, 1993). MPAs allow the development of politically palatable science-management links. It is important that the framework includes the well protected reserve philosophy of the MFRs.

The concept of marine reserves and sanctuaries must be integrated with relevant physical oceanography and life history biology. To protect the genetic integrity of heavily exploited populations, we still need reserves that are completely protected and large enough to maintain natural breeding stocks. While the idea of many small reserves as opposed to few large reserves is controversial in the terrestrial literature, the very openness to larval dispersal that makes the reserve concepts so difficult in marine systems also allows the use of many small reserves. Except for air breathing species, extinctions are rarely a threat in most marine systems, so with the exception of rare habitats such as rocky intertidal and enclosed bays, the need for strict

preservation is less important than in terrestrial systems. This emphasizes the feasibility of the MPA idea (Kenchington and Agardy, 1989).

Sources, sinks, and the size of marine reserves

Recruitment is arguably the most fundamental problem in ecology. Locating the source of the larvae and understanding their transport and recruitment processes is one of the most basic of marine ecological objectives and is the critical issue in determining the size of an effective marine reserve. Certainly the general importance of seed stock is well understood, but a generalized means of identifying such seed 'sources' is not possible because it includes an extremely diverse group of organisms and habitats (Davis, 1989; Quinn *et al.*, 1993). This results in serious problems for generalized definition of marine protected areas because the different species may have sources that range from local to entire ocean basins. In all cases, however, it is obvious that the reproductive processes must be protected. Most marine species employ external fertilization of gametes, a high risk endeavour because dilution effects greatly reduce the probability of successful fertilization. Thus, proximity requirements for adults are often such that they must be within 1 m of each other (Denny and Shibata, 1989), and this probably accounts for aggregations during the reproductive season (Shepherd and Brown, 1993). This implies that relatively non-mobile species must occur at sufficiently high densities to ensure fertilization. This general and important fact immediately imposes severe restrictions on the definition of critical habitats that ensures successful fertilization of gametes. The thinning effects of some fishery management schemes may disrupt these density and aggregation requirements.

Post-fertilization and dispersal processes in marine systems vary from extremely limited dispersal of brooding sessile species to dispersal times ranging from seconds to minutes to hours, days, weeks, and months! This also complicates the definition of critical habitats for species such as many crustacea and echinoderms with larval periods ranging from many weeks to months because they may drift hundreds to possibly thousands of kilometres (Philips *et al.*, 1991; Katz *et al.*, 1994). Certainly this also offers a serious challenge to source and sink modelling because the definition of the important source area is so difficult.

Successful settlement is the other critical component of the dispersal process (Tegner and Dayton, 1977; Pawlik, 1992). The period before the larva becomes physiologically capable of settlement is referred to as the 'precompetent period.' Then even after it is sufficiently developed to settle, it may continue to drift for a long time in a 'competent' phase in jeopardy of predation, thus the length of this period is also critical. Jackson and Strathmann (1981) demonstrate that the critical parameters are the mortality rates, the length of the precompetent period and the ratio of competent/precompetent time. Understanding these parameters is important to understanding the dispersal requirements necessary to describe a protected area properly for a particular species. Unfortunately they are rarely understood, even for well studied species of commercial interest.

The recruitment habitat is another important parameter to define with regard to understanding the 'sink'. Recruitment habitats or nurseries can be important demographic bottlenecks (Lough *et al.*, 1989; Hettler, 1989; Rumrill, 1989). It is interesting to note that many species with the longest precompetent periods also have very specific recruitment habitats that help avoid predation, physical disturbance, physiological stress, etc; Wahle and Steneck (1991) review many papers demonstrating this for benthic crustaceans. It is relevant that most crustaceans do have long precompetent periods, but avoid the early density:fertilization bottleneck by utilizing internal fertilization; and the pericarids, a large group of crustacea also avoid the precompetent phase with brood care.

Many conceptual issues of marine refuges for reef fish are considered in the excellent review by Carr and Reed (1993). They define a harvest refuge as a location of restricted harvesting for the purpose of replenishing exploited populations through larval recruitment. They distinguish these from nature reserves established for the protection of species or habitats, and make a formal effort to quantify the refuge

population size which is a factor of the recruitment potential of both the harvested and refuge populations. The calculation of the refuge recruitment potential is complicated by all the issues discussed above as well as the collective fecundity of the refuge population. They conclude that the size of refuges necessary to sustain a fishery are most importantly determined by the rate of harvest and the rate of production of recruits from the refuge and from the harvested areas. The recruitment rates are determined by larval production and intrinsic (reproductive mode and larval behaviour) and extrinsic (predation, resource availability, currents, etc.) factors affecting larval dispersal (e.g. Carr and Reed, 1993, for fish or Phillips *et al.*, 1991, for spiny lobsters). This is a critical factor defining the number, size, and distribution of refuges and it needs to be evaluated on a case-by-case basis.

Carr and Reed attempt to evaluate source-sink dynamics of larval replenishment by considering four models of dispersal from closed populations (e.g. very limited dispersal), single source populations in which almost all recruitment comes from another outside population, multiple source in which many sources contribute to a large larval pool, and limited distance situations in which larvae have limited dispersal abilities. They restrict themselves to fishes, but this is a useful overview easily extended to invertebrates. Invertebrate examples include brooding species such as many cnidaria, pericarids, echinoderms, etc for closed populations, bay populations such as clams (Ayers, 1956) or many populations of species with extremely long precompetent periods such as spiny lobsters for single source replenishment (note that Carr and Reed refer to almost any situation in which larval replenishment comes from outside sources), barnacles, hydroids and many other opportunistic species with multiple source pools, and abalones (Tegner, 1993), many bryozoans and other species with precompetent ranges of hours to days (see Jackson and Strathmann, 1981 for examples). Migratory species are an important and difficult challenge. The main point is that there is an almost overwhelming amount of biological adaptations and requirements that need consideration for management of refuges. Reserve sizes must be evaluated on these grounds with a case-by-case approach. Certainly this is an area in need of much more research integrating physical oceanography and dispersal/colonization biology.

CONCLUSIONS

Fishing exerts a profound effect on almost all components of associated communities and ecosystems. The most sensitive components are rare habitats that serve as nurseries, and species with low reproductive rates. It appears as though most continental shelf and coastal habitats are already heavily disturbed by fishing impacts of many types. We point out that in almost all cases the situation is so desperate that we cannot afford to wait for more research but must begin strong risk aversion management now. The traditional view is that resource exploitation is a right rather than a privilege and that restrictive management cannot be justified without conclusive evidence of adverse effects. As scientific hypotheses are never proven, only disproven, conservative management is very difficult because exploiters can always point out uncertainties about the causal relationships between exploitation and environmental degradation. Here we advocate a more balanced scientific approach that considers both *Type I* and *Type II* errors and the relative risks of various management alternatives, especially with regard to whether other non-target components of the ecosystems deserve protection. We suggest that the burden of proof more properly lies with the exploiter.

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