IMPACTS OF FISHING GEAR ON MARINE BENTHIC HABITATS

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ABSTRACT

Fishing affects seabed habitats worldwide. However, these impacts are not uniform and are affected by the spatial and temporal distribution of fishing effort, and vary with the habitat type and environment in which they occur. Different fishing methodologies vary in the degree to which they affect the seabed. Towed bottom fishing gears and hydraulic harvesting devices re-suspend the upper layers of the sedimentary habitat and hence re-mobilize contaminants and fine particulate matter into the water column. The ecological significance of these fishing effects has not yet been determined.

Structurally complex habitats (e.g. seagrass meadows, biogenic reefs) and those that are relatively undisturbed by natural perturbations (e.g. deep-water mud substrata) are more adversely affected by fishing than unconsolidated sediment habitats that occur in shallow coastal waters. Structurally complex and stable habitats also have the longest recovery trajectories in terms of the re-colonization of the habitat by the associated fauna.

Comparative studies of areas of the sea bed that have experienced different levels of fishing activity demonstrate that chronic fishing disturbance leads to the removal of high-biomass species that are composed mostly of emergent seabed organisms. These organisms increase the topographic complexity of the seabed and have been shown to provide shelter for juvenile fishes, reducing their vulnerability to predation. Conversely, small-bodied organisms, such as polychaete worms and scavengers, dominate heavily fished areas. Such a change in habitat may lead to changes in the composition of the resident fish fauna. Fishing also has indirect effects on habitat through the removal of predators that control bio-engineering organisms such as algal-grazing urchins on coral reefs. However, such effects are only manifested in those systems in which the linkages between the main trophic levels are confined to less than ten species.

Management regimes that aim to incorporate both fisheries and habitat conservation objectives can be achieved through the appropriate use of a number of approaches, including total and partial exclusion of towed bottom fishing gears, and seasonal and rotational closure techniques. Different management regimes can only be formulated and tested once objectives and criteria for seabed habitats have been defined.

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1. INTRODUCTION

[1] Concerns regarding the effects of fishing on the marine environment have been restricted to the effects of harvesting on those species specifically sought by the fishermen and the by-catches of large fauna, such as cetaceans, birds and reptiles. However, in more recent times, there has been a growing appreciation of the wider and subtle effects that modern levels of fishing pressure exert on the marine ecosystem. These wider ecosystem effects of fishing activities on the marine environment have been projected to new prominence, both through scientific activity and media publicity, such that they are widely considered to be crucial considerations in any future management plans. The so called ‘ecosystem approach’ to fisheries management was highlighted at a meeting held in Montpellier, France, in 1998 that focused exclusively on the ecosystem effects of fishing. This meeting, convened by the International Council for the Exploration of the Sea (ICES) and the Scientific Committee for Oceanographic Research (SCOR), attracted over three hundred delegates and produced a symposium volume with 35 peer-reviewed papers that provide an excellent reflection of the growing importance of this area of research (*ICES Journal of Marine Science*, **57**(3).

[2] It is important to understand what we mean by the ‘ecosystem effects’ of fishing (Jennings and Kaiser, 1998). This can be summarized as:

(i) The effects of fishing on predator-prey relationships, which can lead to shifts in community structure that do not revert to the original condition upon the cessation of fishing pressure (known as alternative stable states).

(ii) Fishing can alter the population size and body-size composition of species leading to a fauna composed of primarily small individual organisms (this can include the whole spectrum of organisms, from worms to whales).

(iii) Fishing can lead to genetic selection for different body and reproductive traits and can extirpate distinct local stocks.

(iv) Fishing can affect populations of non-target species (e.g. cetaceans, birds, reptiles and elasmobranch fishes) as a result of by-catches or ghost fishing.

(v) Fishing can reduce habitat complexity and perturb seabed (benthic) communities.

[3] Points 1-4 are dealt with by other papers in this volume; this paper deals primarily with point 5, but should be considered in the context of the other effects of fishing activities. In recent years the effects of towed bottom-fishing gear on benthic communities and habitats has received considerable media attention in both the trade and popular media press. This has been reflected by a rapid increase in the research effort that has addressed these issues (for reviews see Dayton *et al*., 1995; Jennings and Kaiser, 1998; Kaiser, 1998; Watling and Norse, 1998; Auster and Langton, 1999; Kaiser and De Groot, 2000). In particular, Watling and Norse (1998) and Auster and Langton (1999) give lucid accounts of the ecological significance and ecosystem function of marine habitats and describe the processes that are likely to be affected by fishing activities.

2. FISHING AS AN ECOLOGICAL DISTURBANCE

2.1 Physical disturbance

[4] The majority of seabed (demersal) fishing activity is undertaken in shallow seas on the continental shelf at depths <100 m. However, deep water fishing is an increasingly important sector of the industry and notably occurs around sea mounts at depths >1000 m. Benthic communities within continental shelf environments experience continual disturbance at various scales (Hall, 1994). Large-scale natural disturbances, such as seasonal storms and regular (daily) scouring by tidal currents, form a background against which other smaller disturbances occur,

such as those induced by predator feeding activities (Figure 1). Even the small-scale disturbance effects of individual fauna may have a considerable additive effect on benthic communities, creating a long-term mosaic of patches in various states of climax or re-colonization (Grassle and Saunders, 1973; Connell, 1978). This may contribute to the inherent variability found within marine benthic systems, but can be obliterated by larger-scale physical disturbances such as ice-scour or demersal fishing. It is important to consider the relative scale at which fishing disturbance occurs. Given a similar habitat, very intensive but highly localized fishing disturbance may have fewer ecological implications than less intense, but widespread, fishing disturbance.

[5] In any particular habitat, the associated fauna and flora will presumably have adaptations or life-styles that enable them to persist in that environment. In other words, most communities have an in-built resilience to a certain level of physical disturbance. However, the scale and frequency of physical disturbance events may increase to a point where lasting ecological effects are observed, even against a background of natural disturbance. The additive effects of an entire fishing fleet may reach such a threshold. Shallow-water communities on exposed coastlines are likely to be the most resilient to physical disturbance from bottom fishing (Figure 2). For example, Posey *et al.* (1996) recently demonstrated that even large-scale disturbances, such as hurricanes, have relatively short-term effects on shallow water communities adapted to frequent physical disturbance (but see Rees *et al.*, 1977). However, as habitat stability increases, the relative effects of fishing will also increase, as will the longevity and severity of the ecological effects (Theil and Schriever, 1990; Kaiser and Spence, 1996a; Auster, 1998).

2.2 Species-dependent disturbance

[6] While it is possible to envisage how a bottom fishing gear that is towed over the seabed might lead to alterations in seabed habitat structure, it is perhaps less obvious how reductions in target and non-target fishes or other marine organisms might have consequences for seabed habitat. While the former is termed physical disturbance, the latter is an ecological disturbance that affects interactions that occur between different species. A classic example of species-dependent disturbance is the depletion of sea otters on the western seaboard of North America as a result of human harvesting. Sea otters predate sea urchins that eat kelp. The reduction in sea otter numbers led to an increase in sea urchin numbers which increased the herbivore pressure on the kelp beds leading to their eventual decline (Estes and Duggins, 1985). Such a response is known as a trophic cascade and typically occurs in systems in which the linkages between different components of the system (otters, sea urchins and kelp) are very strong (Kaiser and Jennings, 2001). The interactions within the cascade are between organisms that are assigned to major trophic levels within a system, e.g. predator - herbivore - primary producer; very few cascades involve intermediate trophic levels. In such a case of a significant decrease in the population of a predator or herbivore, there are few others to take their place in the short term. This contrasts sharply with many of the more “open” marine ecosystems, where there may be three or more predators that exert similar levels of predation on one or more species. For example sharks, cetaceans and marine mammals all eat pilchard in the Benguela system off the eastern coast of Africa (Figure 3). Fishing reduces the numbers of large predatory fish, yet the loss of these high level predators can have limited consequences for their prey species. In many of the species rich marine ecosystems there is a high degree of variance in size within the main phylogenetic groupings (containing genetically related species). This is particularly apparent on tropical reefs. Moreover, the phylogenetic groupings tend to contain a high number of species, with a wide range of life history traits, behavioural differences and feeding strategies (Jennings and Kaiser 1998). As a result, predation is diffuse (Hixon, 1991) and while the overall effect of all piscivores on their prey can be substantial the impact of one any individual species, or small group of species, is minor.

3. PHYSICAL DISTURBANCE OF HABITATS BY FISHING ACTIVITIES

[7] Towed bottom fishing gears (trawls, dredges, drags, hydraulic devices) are used to catch those species that live in, on or in association with the seabed. Such gear is designed to catch bottom dwelling species, hence they are intended to remain in close contact in the seabed. The passage of this fishing gear over the seabed can be summarized as follows:

- Disturbance of the upper layers of the seabed causing short-term re-suspension of sediments, re-mineralization of nutrients and contaminants, and re-sorting of sediment particles.
- Direct removal, damage, displacement or death of a proportion of the animals and plants living in or on the seabed.
- A short-term attraction of carrion consumers into the path of the fishing gear.
- The alteration of habitat structure (e.g. flattening of wave forms, removal of rock, removal of structural organisms).

3.1 Effects of re-suspension

[8] The direct physical contact of fishing gear with the substratum can lead to the re-suspension of sediments and the fragmentation of rock and biogenic substrata. The re-suspension, transport and subsequent deposition of sediment may affect the settlement and feeding of the biota in other areas. Sediment re-suspended as a result of bottom fishing will have a variety of effects including: releasing nutrients held in the sediment; exposure of anoxic layers; release of contaminants; increasing biological oxygen demand; and smothering of feeding and respiratory organs. The quantity of sediment re-suspended by trawling depends on sediment grain size and the degree of compaction, and is higher on mud and fine sand than on coarse sand. Transmissiometers, which measure background light levels in water, were reported to frequently recorded the highest levels of turbidity during periods of trawling activity (Churchill, 1989). In deeper water, where storm-related bottom stresses have less influence, otter trawling activity contributed significantly to the re-suspension of fine material. Churchill (1989) calculated sediment budgets for areas of the mid-Atlantic Bight and concluded that trawling was the main factor initiating the offshore transport of sediment at depths of 100-140 m. However, the transport of sediment resulting from fishing activities would not produce significant large-scale erosion over a period of a few years. The effects of sediment re-suspension are clearer in deep-water environments that are relatively undisturbed. Thiel and Schriever (1990) experimentally harrowed an area of seabed at a depth of 4000 m. Their observations revealed that 80% of their study site was covered by fine material that had settled out from the resultant sediment plume. Although this study was designed to imitate the effects of deep-sea mining, the observations are also relevant for deep-sea trawling activities. The observations of sediment re-suspension in the deep sea may resemble the seasonal settlement of organic material that occurs in deep-sea regions (Angel and Rice, 1996).

[9] The sediment/water interface of marine sediments is an important site of benthic primary production. Brylinsky *et al*. (1994) found that benthic diatoms bloomed within otter door tracks one month after they had been created. They reasoned that the bloom was triggered by the release of nutrients from the sediment following trawling. The intensive trawling of *Posidonia oceanica* meadows in the Mediterranean Sea may lead to reductions in littoral primary productivity since large areas of *P. oceanica* are reported to have been killed by the mechanical action of fishing gears and the deposition of re-suspended sediment (Guillén *et al.*, 1994). These meadows are known to be important sources of primary production, although consequences of losses in production are not known. It is unlikely, using existing data, that large-scale changes in primary production could be reliably correlated with changes in fishing intensity.

3.2 Effects of fishing on the habitat

3.2.1 Effects of drive netting, poisons and explosives in the tropics

[10] Techniques such as drive-netting, pull-seining, poison and explosive fishing are principally used by small-scale and artisanal fishers fishing on tropical reefs. Although the effects attributable to the activities of individual fishers are often small in comparison with those attributable to commercial fishing boats using towed gears, the combined effects of their activities are considerable, given the large proportion of the coastal population involved in fishing (Pauly, 1988; Pauly *et al.*, 1989; Dalzell *et al.*, 1996). Many of the fishing techniques used to catch reef-associated fishes cause direct physical damage to the reef substratum. The most widely used destructive fishing techniques are drive netting (Carpenter and Alcala, 1977; Gomez *et al.*, 1987), trapping (Munro *et al.*, 1987) and explosive fishing (Munro *et al.*, 1987). In addition, those poisons widely used to catch fishes for the aquarium trade and consumption have the potential to cause chemical damage to corals and non-target fishes and invertebrates (Rubec, 1986; Eldredge, 1987; McAllister, 1988; Pyle, 1993).

[11] Corals perform several important functions in tropical environments. They provide substrata for primary production, habitats for invertebrates and fishes, and often play a key role in protecting coasts from wave exposure and erosion. The rate at which reefs develop is determined by the balance between rates of accretion owing to the growth of corals, hydrocorals and coralline algae, and erosion owing to mechanical processes and bio-erosion. Fishing affects reefs directly when gears contact the reef substratum or indirectly by altering the relationships between those communities of plants, invertebrates and fishes that determine rates of reef accretion and bio-erosion. Coral accretion relies upon the successful settlement of young corals, and the maintenance of suitable conditions for their growth (Pearson, 1981). These processes may be affected by fishing activities.

[12] Drive netting techniques are used to catch a range of reef-associated fishes, which shelter within the reef matrix or shoal above the reef. These techniques are extensively used on coral reefs, and may range from small-scale village-based operations involving four or five fishers, to large commercial operations that target offshore reefs in the Philippines and South China Sea, and involve hundreds of divers (McManus, 1996). The process of drive netting requires that the fishers (who stand on the reef or dive) scare reef-associated fishes towards an encircling net or trap, using scaring devices such as weighted lines or poles. In shallow water, corals are often broken deliberately to scare closely reef-associated fishes such as groupers (Epinephelinae), snappers or emperors from their refuges. In deeper water, the *Kayakas* and *muro-ami* drive-netting techniques involve teams of swimmers that repeatedly drop weighted scarelines onto the reef in order to drive fishes towards a bag net. Carpenter and Alcala (1977) calculated the damage to one hectare of reef during a single *muro-ami* operation involving 50 fishers who each struck the bottom 50 times with a 4 kg weighted scareline: 6% of the total area of coral present was damaged.

[13] Blast fishing is practised on many reefs in the Atlantic, Pacific and Indian Oceans (Gomez *et al*., 1981; Polunin, 1983; Galvez and Sadorra, 1988; Ruddle, 1996). A variety of explosives are used, including those obtained from mines or removed from armaments. Pelagic fishes living above the reef are often targeted rather than fishes living in direct association with the reef (Saila *et al*. 1993). Owing to the considerable variation in the types and sizes of charges used, and the depths at that they explode, it is difficult to make useful generalizations about the damage that they will cause. Alcala and Gomez (1987) report that a bottle bomb exploding at or near the bottom will shatter all corals within a radius of 1.15 m, and that a gallon-sized drum will have the same effect over a radius of 5 m. A “typical” charge will kill most marine organisms, including invertebrates, within a radius of 77 m. Such techniques are highly unselective and Munro *et al.* (1987) report that post-larval and juvenile fishes are also killed. These young fishes would be about to recruit to the reef habitat, and the repeated effects of blast fishing on a large scale would

reduce fish production from the reef. On those reefs from 15°-30° either side of the equator, which are susceptible to hurricane damage, the effects of blast fishing are often localized and negligible in comparison with those of hurricanes (S. Jennings, personal observation). In other areas, especially in the Philippines, damage attributable to blast fishing is an increasing cause of concern.

[14] Stupefacients are widely used by reef fishers. Traditionally, poisons extracted from plants were extensively used for reef fishing, but in the last few decades, synthetic chemicals such as sodium cyanide and chlorine have been used more frequently (Rubec, 1986; Eldredge, 1987). McAllister (1988) estimated that 150 t of sodium cyanide is used annually on Philippine reefs to catch aquarium fishes. There is little knowledge of the effects of these chemicals on the various life-history stages of the reef biota (Rubec, 1986; Pyle, 1993) and while concentrations of stupefacients that have an acute effect are quickly dispersed, the chronic effects may be significant.

[15] The long-term direct effects of fishing on reefs are largely determined by the rate at which coral can accrete in relation to the rate at which it is damaged. The recovery and recolonization of coral communities following mechanical damage by fishing gears takes place when partially damaged colonies or coral fragments re-grow and when the substratum becomes suitable for coral settlement (Pearson, 1981). Saila *et al.* (1993) developed a model to examine the effects of blast fishing on reefs in the Philippines. At present fishing intensities, the loss of diversity and coral cover would continue for approximately 25 years before recovery is expected. Coral growth rates are highly variable: 0.7 to 17.2 cm/yr for branching species and 0.5-1.9 cm/yr for massive species (Loya, 1976; Huston, 1985; Witman, 1988). Several studies of reef development following hurricanes and other natural events provide a useful guide to recovery rates. Published estimates of recovery time often vary widely because they reflect differences in the authors’ assumptions regarding the organization of coral communities and the meaning of ‘stability’ (Moran, 1986; Done, 1987, 1988; Done *et al.*, 1988; Endean, *et al.*, 1988; Moran, 1990; Turner, 1994; McClanahan *et al.*, 1996). However, a coral community dominated by fast growing branching species and which provides a suitable habitat for many reef fishes would develop within five years (Pearson, 1981).

3.2.2 Effects of towed bottom fishing gear

[16] The short-term effects of fishing on seabed biota are well document in recent studies (for reviews, see Jennings and Kaiser, 1998; Kaiser and De Groot, 2000). The results from short-term studies are informative and often have confirmed our expectations of the type of changes that might occur as a result of fishing activity. Nevertheless, the usefulness of each study on its own is limited by factors such as the specific location, type of gear used and season during which the study in question was undertaken. Viewed on their own, these individual studies can only be used to predict the outcome of fishing activities in a restricted number of situations. However, Collie *et al.* (2000) overcame this problem by extracting summary data from a population of fishing impact studies and undertook a meta-analysis (e.g. Gurevitch and Hedges, 1999) of the combined data set to ask the following questions:

- Are there consistent patterns in the responses of benthic organisms to fishing disturbance?
- How does the magnitude of this response vary with habitat, depth, disturbance type and among taxa?
- How does the recovery rate of organisms vary with these same factors?

[17] Collie *et al.* (2000) found that the magnitude of the immediate response (i.e. change in abundance or biomass) of organisms to fishing disturbance varied significantly according to the type of fishing gear used in the study, the habitat in which the study was undertaken, and among different taxa.

Effects of different gears

[18] The initial impacts of different fishing gears were mainly consistent with expectations. Intertidal dredging activities had a more marked effect than scallop dredging, which in turn had greater effects than otter trawling (Figure 1). Although at first sight the apparent lack of effect from beam trawling is somewhat surprising, it is suspected that the relative paucity of data for this gear is almost certainly part of the explanation. It should also be borne in mind, however, that beam-trawling studies were generally conducted in relatively dynamic sandy areas, where initial effects may be less apparent or are less easily detected. Fishing disturbance effects of intertidal dredging are likely to have the greatest initial effects on the biota because fishers are able to use the harvesting machinery accurately, working parallel lines along the shore. In contrast, fishers using towed nets in subtidal areas are unable to actually see precisely where their gear is fishing, although technological advances in positioning systems are making it increasingly easier to achieve very accurate positioning of fishing gear on the seabed. It is also not insignificant that it is easier for a scientist to accurately collect samples from intertidal compared with subtidal areas, where sampling error is undoubtedly introduced, with potential detrimental effects for the statistical power to detect change. Otter trawling appears to have the least significant impact on fauna compared with other gear, although it is necessary to flag a few warnings about this observation. Firstly, it is the otter doors that hold the wings of the otter trawl open that have the greatest impact on the sediment habitat. However, the otter doors constitute a small proportion of the total width of the gear (ca. 2 m against 40 - 60 m). Secondly, none of the studies published at that time considered the effects of rockhopper otter trawls on seabed communities, and these have been shown to have considerable negative short-term effects on emergent epifauna (Prena *et al.*, 1999; McConnaughey *et al.*, 2000; Pitcher *et al.*, 2000).

Effects in different habitats

[19] Several authors have suggested that the relative ecological importance of fishing disturbance will be related to the magnitude and frequency of background of natural disturbances that occur in a particular marine habitat (Kaiser, 1998; Auster and Langton, 1999). Certainly, it makes intuitive sense that organisms that inhabit unconsolidated sediments should be adapted to periodic sediment re-suspension and smothering. Similarly, it seems plausible that organisms living in seagrass beds rarely experience repeated intense physical disturbances or elevated water turbidity as created by bottom fishing gear (Figure 1). Indeed, such intuition has been the cornerstone of hypotheses about impacts and recovery dynamics for benthos (e.g. Hall, 1994; Jennings and Kaiser, 1998). However, Collie *et al*. (2000) found that their initial impact results with respect to habitat were somewhat inconsistent among analyses. While the initial responses to fishing disturbance of taxa in sand habitats were usually less negative than in other habitats, a clear ranking for expected impacts did not emerge (Figure 4). Such inconsistencies may reflect interactions among the factors arising from the unbalanced nature of the data, with many combinations of gear and habitat absent. For example, the relatively low initial impact on mud habitats may be explained by the fact that most studies were done with otter trawls. If data were also available for the effect of dredgers on mud substrata, a more negative response for this habitat might have been observed. Nevertheless, it should be borne in mind that initial effects of disturbance might be hard to detect in mud communities, which often have low abundances of biota and which tend to be burrowed deep (10 – 200 cm) within the sediment. Presumably, the deeply burrowed fauna would be relatively well protected from the physical effects of disturbance, although the passage of the gear will cause their burrows to collapse. Whether these inconsistencies can be explained in this way can only await further study. It is also important to note that it is important not to classify habitats by the particular nature of the sediment. For example, intertidal sandflats inhabited by high densities of tubiculous worms such as spionids will be more stable (and hence more adversely affected by fishing) than sandflats with relatively little infauna (Thrush *et al.*, 1996).

Immediate effects on biota

[20] Collie *et al*. (2000) found that the most consistently interpretable result within their meta-analysis was the vulnerability of fauna, with a ranking of initial impacts that concurred broadly with expectations based on morphology and behaviour. The lugworm, *Arenicola* spp., had the greatest initial response to disturbance, which is not surprising given that this was the target of a commercial fishery (Figure 4).

[21] Collie *et al*. (2000) also undertook a regression tree analysis that perhaps provides the first quantitative basis for predicting the relative impacts of fishing under different situations. Following the tree from its root to the branches, we can make predictions, for example, about how a particular taxon would be affected initially by disturbance from a particular fishing gear in a particular habitat. Thus, trawling would reduce anthozoa (anemones, soft corals, sea ferns) by 68%, whereas asteroid starfishes would only be reduced by 21%. Similarly, repeated (chronic) dredging is predicted to lead to 93% reductions for anthozoa, malacostraca (shrimps and prawns), ophiuroidea (brittlestars) and polychaeta (bristle worms), whereas a single (acute) dredge event is predicted to lead to a 76% reduction. This approach might ultimately provide a useful quantitative framework for predicting fishing impacts that can be updated and refined as new data emerge.

4. RECOVERY RATES AFTER TRAWL DISTURBANCE

4.1 Soft sediments

[22] From a personal perspective, in an environment that is open to disturbance by fishing gear, the short-term effects of bottom-fishing disturbance on habitats and their biota are of interest, but of far less ecological importance than the issue of the potential for recovery or restoration. Unfortunately, relatively few studies of trawl disturbance have included a temporal component of sufficient duration to address longer-term changes that occur as a result of bottom fishing disturbance. This is almost certainly a result of the conflict between financial resources, project duration, statistical and analytical considerations. Nevertheless, Collie *et al*. (2000) were able to incorporate studies that included a recovery component into their analysis. This permitted them to speculate about the level at which physical disturbance becomes unsustainable in a particular habitat. For example, their study suggested that sandy sediment communities are able to recover within 100 days, which implies that they could perhaps withstand 2-3 three incidents of physical disturbance per year without changing markedly in character (Figure5). This level of fishing disturbance is the average predicted rate of disturbance for the whole of the southern North Sea. However, when fishing effort data is collected at a fine spatial (9 km^2) resolution (Rijnsdorp *et al*., 1998) it becomes clear that effort is patchily distributed and that some relatively small areas of the seabed are visited by >400 trawlers each year. This level of fishing equates to a total disturbance of approximately 8 times per year (Rijnsdorp *et al*., 1998). If our recovery rate estimates for sandy habitats are realistic, this would suggest that these areas of the seabed are held in a permanently altered state by the physical disturbance associated with fishing activities.

[23] At this point, there are some important limitations within the data that should be considered. First, the small spatial scale (the maximum width of most of the disturbed areas examined was <50 m) of most of the trawl impact studies make it likely that much of the re-colonization was via active immigration into disturbed patches rather than reproduction within patches. The authors found recovery to be slower if the spatial scale of impact was larger, as it would be on heavily fished grounds due to the additive effects of an entire fleet of trawlers. Second, it should be noted, that while one might accurately predict the recovery rate for small-bodied taxa such as polychaetes, which dominate the data set, sandy sediment communities often contain one or two long-lived and therefore vulnerable species. Note, for example, the occurrence of the large bivalve *Mya arenaria* in the intertidal zone of the Wadden Sea. While the majority of the benthos in this environment recovered within 6 months of lugworm dredging, the biomass of *M. arenaria*

remained depleted for at least 2 years (Beukema, 1992). This delayed recovery of larger-bodied organisms is no doubt even more important in habitats that are formed by living organisms (e.g. soft corals, sea fans, mussels) as the habitat recovery rate is directly linked to the re-colonization and growth rate of these organisms. By now, there is sufficient evidence in the literature to suggest that under conditions of repeated and intense bottom-fishing disturbance, a shift from communities dominated by relatively high biomass species towards dominance by high abundances of small-sized organisms will occur.

4.2 Biogenic habitats

[24] It is clear that intensively fished areas are likely to be maintained in a permanently altered state, inhabited by fauna adapted to frequent physical disturbance. These effects will be most apparent for stable types of habitats that contain structural biogenic components. Presumably, such habitats will have the longest recovery time compared to less stable substrata. Yet it is for these habitats that the paucity of data is most apparent. While it would appear that none of the habitats included in Collie *et al.*'s (2000) study fall into this category, some new data are beginning to emerge. Hall-Spencer and Moore (2000) examined the effects of fishing disturbance on maerl beds. Maerl beds are composed of highly dichotomous calcareous algae. This forms a complex substratum with a high degree of 3-dimensional complexity. Not surprisingly, the associated assemblages have high diversity and many of the associated species are large-bodied and slow-growing. Hall-Spencer and Moore (2000) showed that four years after the occurrence of an initial scallop-dredging disturbance had occurred, certain fauna, such as the nest building bivalve *Limaria hians,* had still not re-colonized trawl tracks. Similarly, work by Sainsbury *et al.*, (1987, 1999) suggests that recovery rates may exceed fifteen years for sponge and coral habitats off the western coast of Australia. The presence of such habitats was important for fish species of commercial importance.

5. DEDUCING THE EFFECTS OF CHRONIC DISTURBANCE

[25] The perceived problems that might be associated with intense and prolonged bottom-fishing disturbance have only been examined in the last 20 years. However, the bottom-fishing fleets have been in operation much longer. For example, the beam trawl fleet in the southern North Sea expanded dramatically through the 1960s and 1970s. Consequently, many present-day studies have been undertaken in what is already a considerably altered environment. Despite our efforts to predict the outcome of fishing activities for existing benthic communities, we are often unable to deduce the original composition of the fauna because data gathered prior to the era of intensive bottom fishing are sparse. This is an important caveat, because recent analyses of the few existing historical datasets suggest that larger-bodied organisms (both fish and benthos) were more prevalent prior to intensive bottom trawling (Greenstreet and Hall, 1996; Frid *et al.*, 2000; Rumohr and Kujawski, 2000). Moreover, in general, epifaunal organisms are less prevalent in areas subjected to intensive bottom fishing (Collie *et al.* 1997; Sainsbury *et al.*, 1997; Kaiser *et al.*, 2000a, b; McConnaughy *et al.*, 2000; Rumohr and Kujawski, 2000). An important consequence of this effect is the reduction in habitat complexity (architecture) that accompanies the removal of sessile epifauna. Nevertheless, it has been hard to convincingly demonstrate that towed bottom fishing activity has been responsible for changes in bottom fauna and habitats. Often, effort data are lacking at a scale or over a time period that is relevant to ascertain the disturbance history of a particular area of seabed. In the few instances when such data has been available, observations have indicated consistently a shift from dominance by high biomass organisms towards communities dominated by small-bodied opportunistic species (Collie *et al.*, 1997; Engel and Kvitek, 1998; Bradshaw *et al.*, 2000; Kaiser *et al.*, 2000a, b; McConnaughey *et al.*, 2000). It is becoming increasingly apparent that habitat modification appears to have important consequences for fish communities (Sainsbury *et al.*, 1987; Auster and Langton, 1999; Kaiser *et al.*, 1999).

6. INDIRECT EFFECTS OF FISHING

6.1 Interactions that result from exploitation of target species on tropical reefs

[26] The overexploitation or effective conservation of fishes on tropical reefs can affect community structure and a range of ecosystem processes. Many reefs are intensively fished since they provide the main sources of protein and income for coastal people with few other opportunities for fishing, farming or hunting. Reef fishers target species from all trophic groups, and on many fished reefs, the abundance of herbivorous and invertebrate feeding fishes has been reduced by an order of magnitude or more (Russ, 1991).

[27] The main groups of algal consumers on reefs are herbivorous fishes and sea urchins. The abundance of sea urchins is regulated by recruitment success, food supply and natural mortality due to predation and disease. The main urchin predators are fishes such as emperors (family Lethrinidae) and triggerfishes (Balistidae) that are also targeted by fishers (McClanahan, 1995b). On some reefs in the Caribbean and East Africa, fish predation appears to play a key role in controlling the abundance of urchin populations and they have proliferated following the overfishing of their predators (McClanahan and Muthiga, 1988; McClanahan, 1992, 1995a).

[28] Once urchins have become abundant, they graze the majority of algal production. Urchins can tolerate low algal biomass because they have low consumption and respiration rates. This allows them to outcompete herbivorous fishes that have higher consumption and respiration rates and reach maximum biomass levels an order of magnitude higher (McClanahan, 1992). Since the herbivorous fishes are poor competitors, they may not recover to former levels of abundance when fishing is stopped (McClanahan, 1995a).

[29] As they graze, urchins erode the reef matrix and prevent the settlement and growth of coral recruits. Unless recruitment failure or urchin disease leads to a collapse of urchin populations, other intervention is needed to promote recovery of the reef ecosystem. McClanahan *et al*. (1996) attempted such intervention on a small scale by deliberate removal of urchins. When they removed urchins from unfished experimental plots on Kenyan reefs, there were significant increases in algal cover and fish abundance within one year. However, on fished reefs, herbivorous fishes were less abundant, and the algae rapidly overgrew corals as they proliferated. The ecosystem shifts that McClanahan *et al*. (1996) induced in fished areas by urchin removal were remarkably similar to those observed in the heavily fished Caribbean when there was mass urchin mortality following disease. Here, the loss of urchins led to heavy growths of algae that soon dominated the reef community (Carpenter, 1985; Lessios, 1988).

[30] The effects of predator removal on urchin populations contrast with the effects of piscivore removal on reef fish populations. While many studies have shown that the abundance of piscivorous reef fishes is dramatically reduced by fishing, there is little evidence for a corresponding increase in the abundance of their prey. We will give some examples of this effect, and consider why the response of prey fish communities is so weak.

[31] Several studies have documented significant decreases in the abundance of piscivorous target species following fishing, and yet there was no evidence for a corresponding increase in the abundance of their prey (Bohnsack, 1982; Jennings and Polunin, 1997; Russ, 1985). The reasons for this are likely to be linked to the structure of reef fish communities, where phylogenetic groupings contain many species, with a wide range of life history traits, behavioural differences and feeding strategies (Hiatt and Strasburg, 1960; Hixon, 1991; Parrish *et al.*, 1985, 1986). Moreover, most fish species undergo marked ontogenetic changes in diet, and act as the prey and predators of other species in the course of their life history. As a result, while the collective impacts of predators are large, the impacts of individual predator species on the dynamics of their prey are minor. This effect was termed diffuse predation by Hixon (1991).

[32] It is worth noting that on much smaller scales (m^2 rather than km^2) there is some evidence for the role of predation as a structuring force, particularly when habitat or refuge space is directly limited. Thus Caley (1993), Hixon and Beets (1993) and Carr and Hixon (1995) have conducted elegant studies that demonstrated that experimental reductions in piscivore abundance lead to detectable decreases in the abundance and diversity of their prey. However, even at these scales, it is widely accepted that recruitment variation has a more significant impact on population structure (Doherty, 1991; Doherty and Fowler, 1994; Sale ,1980).

6.2 Interactions following removal of predatory fish in temperate marine fisheries

[33] Fishing has a marked impact on the dynamics of exploited fish populations. Many species, both target (caught intentionally) and non-target (caught as by-catch), can sustain fishing mortality rates 2-3 times their natural mortality rates, but are ultimately depleted as fishing mortality rises. Even in the most intensively fished marine ecosystems, such as the Benguela upwelling, Georges Bank, Bering Sea and North Sea, piscivorous fish are more important consumers of fish biomass than humans, marine mammals or birds (e.g. Bax, 1991; Yodzis, 1994). Long-term studies of fish populations have suggested that the intensive fishing of predators has allowed other species to proliferate. For example, (Sherman *et al*., 1981) showed that population explosions of sandeels in the northeast and west Atlantic coincided with the depletion of their predators, such as Atlantic herring *Clupea harengus* and mackerel *Scomber scombrus*. However, it is very difficult to establish causality given the range of environmental factors that could also account for these links.

[34] In a recent review of the effects of predator depletion on non-target species in temperate marine ecosystems, Jennings and Kaiser (1998) concluded that there were relatively few circumstances in which changes in the abundance of piscivorous fishes in marine ecosystems had cascading impacts on other parts of the system. High fish consumption rates did not consistently imply that predation was a structuring force when most species can act as predators and prey in the course of their life history and when adult predators are still capable of switching diet and feeding strategy in response to prey availability. Indeed, in temperate systems, the strongest evidence for the predator-based control of prey species came from the impact of humans on their target species. The strength of this relationship is likely to result from the conservative fishing strategies employed by humans, who, in the majority of commercial fisheries, are unwilling to be flexible in their aims and target a relatively small proportion of the total fish fauna. Most predatory fish, conversely, are very generalist feeders, often switching to invertebrate prey or cannibalism and eating many species of fishes at different stages in their life history.

7. ESSENTIAL FISH HABITAT

[35] Recent amendments to the USA *Magnuson-Stevens Act* require fisheries managers to define ‘essential fish habitat’ (EFH) and address the impact of fishing gear in their management plans (Benaka 1999). This is probably one of the first legislative steps taken in fisheries management that will require the assimilation and application of the scientific knowledge outlined in the paragraphs above. In many ways, this legislation is one of the first measures to embrace an ecosystem perspective in fisheries management. In some instances, it is fairly simple to identify those habitats that might be considered essential to the life-history of some species. Such habitats include spawning and nursery areas, many of which are protected from fishing activity in European waters. However, of equal relevance are the habitat quality issues that affect the acquisition of food and the avoidance of predators. Hence there is an urgent need to identify those habitats that have an important or ‘essential’ functional role for particular species or types of fish (e.g. piscivores/herbivores/omnivores or flatfish/roundfish) at other stages of their life history.

[36] Previous studies of the relationship between fish and shellfish assemblages and their environment have focused on variables such as salinity, depth and substratum type (e.g. Smale *et al*., 1993; Overholtz and Tyler, 1985). Yet, while such environmental parameters are in some

cases good correlates of certain fish assemblages, they do not necessarily define the essential features of a specific habitat, rather they constitute a component of that habitat. Habitat complexity and composition (e.g. grain size composition) appear to be important physical features for some fish species (e.g. Sainsbury, 1987; Gibson and Robb, 1992). Many studies have already demonstrated the relationship between flatfish species and the sediment particle composition of the seabed, which may be more important than the occurrence of associated epibenthic structures or fauna that occur in that habitat (e.g. Gibson and Robb, 1992; Rogers, 1992). Hence, a specific particle size-composition may be essential for flatfish, whereas the presence of large sessile epifauna or rocky substrata might be considered non-essential. In contrast, there is good evidence to suggest that structural complexity can have important implications for the survival of roundfishes (e.g. Walters and Juanes, 1993).

[37] Habitat complexity is a product of the surface topography of the substratum and the sessile epifauna that grow upon it. Reef-forming organisms can result in habitats of very high complexity, providing a multitude of refuges for a diverse range of species. More subtle features such as sand ridges and pits created by the feeding or burrowing action of benthic fauna may provide shelter for bottom-dwelling fish species (e.g. Auster *et al.*, 1997). Bottom fishing activities are capable of greatly reducing habitat complexity by either direct modification of the substratum or removal of the fauna that contribute to surface topography (Auster and Langton, 1998; Jennings and Kaiser, 1998). Hence, degradation of habitat complexity by fishing activities may lead to changes in the associated fish assemblages (e.g. Sainsbury *et al*., 1997). Alteration of habitat features has been shown to have important consequences for freshwater fishes, and this is the caveat that underpins much of the ecological restoration projects centred on salmonid habitats. An initial study of habitat/fish assemblage relationships indicated that even subtle alterations in habitat characteristics can cause a shift in the dominance of certain fish species within the assemblage (Kaiser *et al*., 1999). Presumably, a good understanding of the link between fish and their habitat would enable us to predict the consequences of habitat alteration. For example, for certain species such as sole *Solea solea* that preferentially live in relatively uniform sandy areas, the exclusion of towed bottom fishing gear from an area of the seabed could permit the growth of emergent sessile fauna that make the environment better suited to predatory flatfishes such as plaice *Pleuronectes platessa* and dab *Limanda limanda* (Kaiser *et al*., 1999). Thus, in the case of the sole fishery, the fishing activity may maintain the seabed habitat in a condition that favours the target species. Quite clearly the opposite would be true for any species of fish favouring more complex habitats.

8. INTEGRATING HABITAT CONSERVATION OBJECTIVES INTO FISHERIES MANAGEMENT

[38] It would appear that with sufficient scientific information it should be possible to formulate a regime of fishing effort (= physical disturbance for towed bottom-fishing gear) that would be environmentally sustainable. Here we define environmentally sustainable as the process by which the habitat and its associated biological assemblage can recover before a subsequent disturbance event. For example, in shallow sandy areas of the seabed, two to three physical disturbances of the seabed every year may have little or no net effect on the habitat or resident assemblage. However, at present, the definition of sandy areas is too imprecise a habitat criterion on which to base such a management plan. We know, for example, that sand flats that are dominated by tube-building spionid worms take much longer to recover if these worms are removed through physical disturbance, as the worms normally have a stabilizing effect on the habitat (Thrush *et al*., 1996). Nevertheless, the complete exclusion of bottom-fishing disturbance from sandy habitats that are fished at present may actually have a negative effect on the fishery, as suggested in Section 7. Physical disturbance will, to some extent, promote dominance by opportunistic species such as small polychaetes that form a major component the ecosystems of many commercially important flatfish species (Rijnsdorp and van Leeuwen, 1996).

[39] What is clear from the studies undertaken to date is that there exist communities and habitats that are so sensitive to physical disturbance that all forms of bottom-fishing with towed gear should be considered for exclusion from these areas forthwith. As a matter of urgency, there is need to identify other habitats that have long recovery times and that are exposed (or might in the future be exposed) to towed bottom-fishing gear – the most likely candidates are those that contain a high proportion of structural fauna. In European waters examples of such habitats would include:

- Deep sea coral reefs of *Lophelia pertusa.*
- Maerl beds.
- Reefs of mussels (*Modiolus modiolus*) and *Limaria hians.*
- Areas of the seabed with aggregations of sea fans.
- Beds of fan mussels (e.g. *Atrina fragilis*).
- Sea grass meadows.

[40] It is important at this point to define what we mean by sensitive fauna or habitats. Sensitive fauna may be defined by their physical attributes (e.g. fragility of body structure), their reproductive strategies (e.g. infrequent recruitment or low reproductive output) or remaining population size (e.g. the lower the population size, the more vulnerable to extinction that species will be). Sensitive, non-sensitive, structured and non-structured fauna and habitats – all will be affected to some degree by towed bottom-fishing gear. However, for effective management, we need to define and identify vulnerable species or habitats and the management approach(es) that should be used. This usually requires clearly defined management objectives (that can be measured and monitored) and data on fishing effort (level and spatial distribution), impact and recovery times. From this information, management strategies can be developed and tested against the objective(s). We can then start defining what we mean by a sustainable fishery.

9. FISHERY MANAGEMENT MEASURES THAT INCLUDE HABITAT MANAGEMENT

[41] Fisheries management that includes habitat management as an objective could be achieved through a number of different mechanisms. Total exclusion of all fishing effort will achieve habitat conservation, provided that there are no other extrinsic factors that negatively affect that habitat (e.g. agricultural run-off from adjacent land masses). Other measures include networks of area closures at different spatial scales that are determined by the demography of the species to be protected. Temporal closures may achieve habitat preservation if the habitat or species in question have generation times that fall within the time-scale of the temporal closure.

[42] The total exclusion of certain forms of fishing activities from areas of the seabed will inevitably lead to opposition from the fishing industry, and by its nature, such measure are extreme. Nevertheless a recent large-scale study on the NW Atlantic coast of North American has demonstrated elegantly the effectiveness of such large-scale closures. Alternatively, inshore fisheries lend themselves to the partitioning of seabed resources such that certain areas of the seabed can continue to be exploited using gear that cause minimal environmental damage. The two following examples illustrate the potential of such approaches.

[43] In New England, USA, seasonal closed areas have been an important component of fisheries management since the early 1970s, but had little impact on the groundfish stocks that they were designed to protect. In 1994, three large areas that in total covered 17 000 km^2 of the seabed were closed all year to all fishing gears that might retain groundfish (trawls, scallop dredges, hooks, etc.). These closed areas were maintained for five years and were found to effectively protect the more sedentary components of the assemblage, such as flatfishes, skates and scallops. Although less protection was afforded to cod *Gadus morhua* and haddock *Melanogrammus aeglefinus,* additional legislation to protect specific important juvenile habitat

lowered stock-wide mortality rates. Scallop dredgers were excluded because they took a by-catch of groundfish species. The relaxation of fishing effort on scallops had dramatic effects and led to a 14-fold increase in scallop biomass within the closed areas during 1994-1998 (Figure 6). A portion of the closed areas was re-opened to scallop dredging in 1999 (Murawski *et al*., 2000). The returns of scallops during this period were so encouraging that managers are now contemplating a formal 'area rotation' scheme for this fishery, presumably on a time-scale of 4-5 years.

[44] The second example comes from an inshore fishery off the south coast of England. When two commercially important species co-exist in the same habitat, conflict may arise between different sectors of the fishing industry. A good example of this situation is when fishers using towed bottom-fishing gears (scallop dredges, beam trawls and otter trawls) operate in the same areas in which fixed bottom gear (crab pots) are deployed. Kaiser *et al*., (2000b) examined an area subject to a voluntary agreement between these two sectors of the fishing industry such that some areas are used exclusively by fixed-gear fishers, some are shared seasonally by both sectors, and others are open to all methods of fishing year-round. This agreement was enacted to resolve conflict between the two sectors of the industry. An additional perceived benefit of this agreement was the possible protection of the seabed from towed bottom-fishing gear. Kaiser *et al*., (2000b) undertook comparative surveys of the benthic habitat and communities within the area covered by the agreement and compared different areas subjected to a range of fishing disturbance regimes. Communities found within the areas closed to towed fishing gears were significantly different from those open to fishing either permanently or seasonally. Abundance/biomass curves plotted for the benthic fauna demonstrated that the communities within the closed areas were dominated by higher biomass and emergent fauna that increase habitat complexity (Figure 7). Areas fished by towed gear were dominated by smaller-bodied fauna and scavenging taxa. While it would appear that gear restriction management regimes have the added benefit of conserving habitats, target species and benthic fauna within the management area, it is at present not possible to determine whether there are any wider benefits for the fishery that exploits the target species outside the management area.

10. FUTURE RESEARCH PRIORITIES

[45] With respect to the design of future studies, we feel that experimentalists wishing to address the fishing impacts issue would be best served by abandoning short-term, small-scale pulse experiments. Instead, the scientific community should be arguing for support to undertake much larger-scale press and relaxation experiments. One half of the experiment has already been done – since fishing activity has been providing the press for many years; what we now require are more carefully designed closed-area contrasts. There clear advantages to this approach. First, the results obtained are clearly interpretable in terms of real-world intensities of fishing disturbance. Second, the spatial scale of the protected areas could be relatively small (and hence replicated, to fulfil the requirements for sound experimental design) without compromising unduly the interpretation of recovery dynamics: estimates of recovery in small protected areas in a sea of disturbance are likely to be conservative, while recovery in small, deliberately disturbed patches are not. Third, the experiments would be conducted in the very habitats (i.e. real fishing grounds) concerning which the question of recovery is actually being posed.

[46] Our current understanding of the functional role of many of the larger-bodied long-lived species is limited and should be addressed to predict the outcome of permitting chronic fishing disturbance in areas where these animals occur. In addition, our understanding of the ecosystem services that many of these species provide is limited by a paucity of scientific understanding (e.g. Hall-Spencer and Moore, 2000).

[47] To date, the majority of studies that have addressed fishing impacts on the seabed have concentrated on the biota, with little reference to gear/sediment interactions (Pilskaln *et al*., 1998).

As fishing gears disturb soft sediment they produce sediment plumes and re-mobilize previously buried organic and inorganic matter. Presumably this increases the release of nutrients into the water column and has important consequences for rates of biogeochemical cycling. To date, this issue has received little attention, with the exception of one or two as yet unpublished studies.

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Figure 1. The relative recovery rates of different scales of disturbance that occur in the marine environment. The figure shows various forms of fishing activity compared with naturally occurring disturbances such as predation effects and physical sources of disturbance (after Hall, 1994).

Figure 2. A simplistic model to illustrate the relative importance of a constant level of fishing disturbance (given an arbitrary value of 20 on the y-axis) in different habitats (habitat stability) that are subjected to different frequencies or levels of natural disturbance. As levels of natural disturbance decline, fishing disturbance accounts for a greater proportion of the total disturbance experienced and becomes increasingly important (adapted from Jennings and Kaiser, 1998).

Figure 3. An example of a marine system
(a) in which trophic cascades are known to occur as a result of the strong linkages between the main components of the system compared with an over-simplified perception of a system in which cascades might be seen to occur (b) whereas no such responses are actually apparent due to the actual complex nature of that system (adapted after Yodzis, 1994)

Figure 4. The predicted mean response derived from ANOVA of
(a) the response of different taxa to physical disturbance that occurred immediately after that disturbance had occurred;
(b) the response of invertebrate abundance or biomass in different habitats;
(c) the response of invertebrate abundance or biomass to different gear types.
Data are on a transformed scale where values correspond to % declines from controls as follows:
-0.1 = 10%; -0.22 = 20%; -0.35 = 30%; -0.5 = 40%; -0.68 = 50%, -1.35 = 75%; -4.61 = 100%.
In all cases, the initial response of the fauna was negative.

Figure 5. Results from a meta-analysis of the effects of fishing disturbance on benthic communities. The scatter plots of the relative change of all species (each data point represents the relative abundance of a different species on each different sampling date) in different habitats at time intervals after the occurrence of a fishing disturbance. The fitted curves show the predicted time trajectory for recovery to occur. On the y-axis, 0 shows no relative change in abundance; negative values show a relative decrease in abundance.

Figure 6. Standardized abundance of sea scallops (numbers per dredge tow) by shell height, taken in the July National Marine Fisheries Service dredge survey on Georges Bank. Data are presented separately for the areas closed and those open to scallop dredging. Harvestable animals are indicated by the 50% selection line (from Murawski *et al.*, 2000).

Figure 7. Abundance/biomass curves of samples collected from areas protected from towed bottom-fishing gear (low disturbance), areas open seasonally to towed bottom-fishing gear and those areas that are fished all year with towed bottom-fishing gear (high disturbance). As the level of bottom fishing disturbance increases, the biomass curve (B) converges with the abundance curve (A), which is a typical response in stressed communities (adapted from Kaiser *et al.* 2000b).