Impacts of Mobile Fishing Gear: The Biodiversity Perspective

ELLIOTT A. NORSE

Marine Conservation Biology Institute
15806 NE 47th Court, Redmond, Washington 98052-5208, USA

LES WATLING

Darling Marine Center, University of Maine, Walpole, Maine 04573, USA

Abstract.— The increasing concern about impacts of bottom trawling, scallop dredging, and other mobile fishing methods has focused primarily on effects on commercial fisheries, but these fishing activities also act more broadly on benthic biological diversity. Because the seabed is erroneously envisioned as a featureless, nearly lifeless plain, impacts of commercial fishing gear have long been underestimated. Structures on and in the seabed, including biogenic structures (reef corals, kelp holdfasts, shells, tubes, and tunnels), create a diversity of habitat patches. They provide refuges from predation and feeding places for demersal fishes and other species. Benthic structural complexity is positively correlated with species diversity and postsettlement survivorship of some commercial fishes. Mobile fishing gear disturbs the seabed, damaging benthic structures and harming structure-associated species, including commercially important fishes, although some other commercial fish species can persist where seabed structures have been removed. Bottom trawling is therefore similar to forest clear-cutting, but it is far more extensive and is converting very large areas of formerly structurally complex, biologically diverse seabed into the marine equivalent of low-diversity cattle pasture. In contrast with the U.S. National Forest Management Act, which governs use of living resources in federally owned forestlands, the 1996 Magnuson-Stevens Fishery Conservation and Management Act does not prevent ecosystem "type conversion" and ignores the need to maintain biological diversity. Preventing further loss of marine biodiversity and key fisheries will depend on our willingness to protect marine areas from effects of mobile fishing methods.

I'd like to be under the sea in an octopus' garden with you.—Beatles (rock group), "Octopus' Garden" (song)

The ocean is a desert with its life underground and the perfect disguise above.—America (rock group), "A Horse with No Name" (song)

Human activities are rapidly reducing the Earth's biological diversity (the diversity of genes, species, and ecosystems). Since Myers (1979), Lovejoy (1980), and Norse and McManus (1980) revealed that the loss of biological diversity is a massive worldwide phenomenon, especially in tropical forests, studies—including the most comprehensive biodiversity status report to date (Heywood 1995)—have found that physical destruction of ecosystems is the most pervasive cause of biodiversity loss. But such studies have focused mainly on land. In the sea, overfishing and pollution have long been considered leading threats to biodiversity, but more comprehensive examination shows that these are only two of five major threats (Norse 1993); one of the other threats is physical destruction of ecosystems. In the sea the leading cause of ecosystem destruction is use of mobile fishing gear such as bottom trawls and dredges, which smooth, crush, and uproot benthic structures (Auster and Langton 1999, this volume; Watling and Norse 1998). To help read-

ers gauge the impacts of mobile fishing gear, we first discuss the importance of seabed structures and the relationship between habitat structural complexity and biodiversity. We then discuss management approaches designed to maintain and restore marine biological diversity.

People who study conservation in benthic ecosystems have much to learn from forests, which cover 38×10^6 km^2 (Perry 1994)—about 7.5% of the Earth's surface—because forests are the best-studied wildlife habitat. Many of the canonical findings in wildlife biology, conservation biology, and landscape ecology, such as the realization that island biogeography theory applies to ecosystems that have been fragmented (Terborgh 1974; Diamond 1975), have come from studies of organisms in forests. In contrast, the seafloor of the world's continental shelves, which cover 28×10^6 km^2 (Sharp 1988), is far less familiar to the human species. Indeed, even many marine scientists rarely or never see the seafloor in person. In the minds of the public, the seabed is ei-

ther a lush garden or a desert, but the truth is more subtle and complex. Marine ecosystems dominated by large (>1 m), living three-dimensional structures, including kelp forests, mangrove forests, and coral reefs, are actually a very small (albeit very important) part of the marine realm (for example, coral reefs occupy only 0.6×10^6 km^2 or 0.1% of the Earth's surface [Reaka-Kudla 1997]). The rest of the seabed that the public sees consists mostly of sandy beaches and muddy plains and may appear featureless to the untrained eye. This perception is mainly a function of people's peculiar perspective; the narrow bands of sandy beaches that people visit are pounded by waves that prevent the growth of most biogenic structures (e.g., sponges, clam shells, amphipod colonies, cerianthid anemone tubes, polychaete worm tunnels, sea cucumber fecal deposits), while the vastly greater areas covered by mud (primarily silt- and clay-sized particles) are home to structures that are often too small to be resolved by cameras towed meters above the seabed. Yet, no less than on land, structures on and in the seafloor are crucial habitat features for most of the world's marine species. Coral reefs alone host 25% of the world's marine fish species (McAllister 1991).

People who study or manage species in the terrestrial realm have long known that avoiding loss of exploited wildlife populations goes beyond limiting mortality through bag, season, and size limits. It is at least as important to maintain features in the habitat such as cover and food that are essential to species' reproduction and growth. Wildlife biologists and other conservation biologists have built a substantial understanding of the three-dimensional spatial structure needed by species from leopards to spotted owls. For example, in the U.S. Pacific Northwest, structurally complex late-successional forests provide the myriad kinds of spaces—holes, cavities, chimneys, overhangs, thickets, lookout posts, and bridges—that many wildlife species need. The clearcuts and tree plantations (i.e., even-age monocultures of trees that are created after ancient forests are clear-cut) that replace them are structurally far simpler and cannot support many of the species found in ancient forests (Norse 1990).

Intuitive understanding of the importance of structural complexity underlies much of the scientific and public concern about clear-cutting, but there is also ample scientific evidence. Nearly four decades ago, MacArthur and MacArthur (1961) pointed out that songbird diversity is higher in forests that are more structurally complex. On land it is not difficult to observe the relationship between structure, which wildlife biologists call "cover," and species that need it, and studies of structure–diversity relationships are now quite sophisticated (e.g., Hansen et al. 1995). In the sea, however, scientific knowledge has depended far less on direct observation than on remote sampling (often using fishing gear) from the decks of vessels. This sampling practice has tended to limit understanding to structure-formers that come up in sampling gear. Fishery biologists have long known that kelp forests, coral reefs, and rocky reefs attract many commercially important fishes and the species they eat. Yet, as Thrush et al. (1998:876) note, "Fishery models often fail to include the potential role of interactions between habitat features and the survivorship of juveniles of exploited stocks." Fishery biology (in contrast to ichthyology and benthic ecology) has been slow to appreciate the importance of small structures on the seabed as habitat and the consequences of their destruction.

The diverse smaller structures of the seabed include cobble- and pebble-sized rocks; sand ripples; thalassinid crustacean mounds; sea cucumber fecal deposits; pits left by feeding rays and crabs; sea-grass blades; the spines of living sea urchins; kelp holdfasts; sponge, sea pen, and bryozoan colonies; many kinds of tunnels; and annelid worm, amphipod crustacean, vermetid gastropod, and cerianthid anemone tubes. These structures are naturally abundant in most marine ecosystems (see synopses in Gage and Tyler 1991 and Giere 1993). Seabed structures can result from past events (e.g., cobbles deposited by melting glaciers) or from ongoing processes (e.g., reef-building by mytilid mussels and sabellariid polychaetes). Some of the most important structures occur below the sediment–water interface, riddling the seabed with a complex of tunnels and tubes (summarized in Wheatcroft et al. 1990). Other structures, ranging from polychaete worm and amphipod crustacean tubes to corals and kelps, reach millimeters to tens of meters into the water column. Although seabed structures are generally far smaller than the ones in terrestrial forests, they are at least as important as habitat features for a myriad of species, including postsettlement young of commercially important fishes. Because structural complexity is so vital in benthic ecosystems, reduction of complexity affects all aspects of benthic biological diversity, including fisheries. Of the many natural and anthropogenic factors that disturb the seabed and reduce structural complexity, the leading factor is fishing with mobile gear (Watling and Norse 1998).

Importance of Seabed Structures:
A Fish-Eye View

Why is seabed structure so important? Biological activity is most pronounced at interfaces, and the interface between the water column and the seabed is no exception. The species diversity and biomass of life in the half-meter above and below the sediment–water interface are usually orders of magnitude higher than in the overlying several meters of the water column (this has long been recognized by paleontologists who study ancient benthic communities [Ausich and Bottjer 1982]). Not only does the seabed collect the rain of detrital particles from above, but it also has the three-dimensional lithic and biogenic structures that provide habitat for innumerable species. Thus, these structures—even ones as small as one or a few centimeters in size—provide cover and food for invertebrates and fishes that eat them. Virtually everybody who has watched marine animals has observed that juvenile and adult fishes, crabs, lobsters, and octopuses stay close to rocks and hide in holes or interstices between rocks when potential predators approach (Bohnsack 1991). As Ebeling and Hixon (1991) noted, without the shelter that complex structures provide, juvenile fishes are highly vulnerable to predators in both tropical and temperate reef ecosystems. Postsettlement Atlantic cod *Gadus morhua*, for example, show strongest survivorship on rugose bottoms (Gotceitas and Brown 1993). Lithic features such as boulders, rock ledges, and sand waves also play important roles in feeding. For example, juvenile red hake *Urophycis chuss* hover just downcurrent of sand wave crests, where they catch zooplankton carried by bottom currents (P. J. Auster, National Undersea Research Center for the North Atlantic and Great Lakes, personal communication), much as trout hover in the lee of sunken logs, catching stream drift. Moreover, the troughs between sand ripples and the pits dug by infaunal-feeding rays and crabs often accumulate organic material and become feeding places for detritivores and their predators. In the Gulf of Maine, areas not frequently disturbed by mobile fishing can have large numbers of redfish *Sebastes fasciatus*, each individual occupying space near the bottom of individual boulders (Auster, personal communication; L. Watling, personal observations).

In the marine realm, the relationship between habitat structural complexity and biodiversity has been best documented for fishes in coral reefs, where structures are conspicuous and direct observation is comfortable for divers. Ormond and Roberts (1997:233) noted, "There is often, for example, a striking relationship between fish species richness and habitat structural complexity or heterogeneity" and went on to note that "such a relationship is well known from terrestrial…as well as other marine studies," although this relationship might not always be strong in coral reefs. In perhaps the earliest study that quantified this relationship in the sea, Risk (1972) found higher fish species richness as coral rugosity increased. In the Tuamotu Archipelago in the South Pacific, Bell and Galzin (1984) found that slight changes in live coral cover resulted in dramatic increases in fish species diversity. Some 68% of the 115 fish species investigated were found only at sites with some live coral. Following the loss of live coral cover on the reefs of the island of Okinawa due to an outbreak of the sea star *Acanthaster planci*, Sano et al. (1984) were able to predict the subsequent loss of fish species. Some species were coral polyp feeders and so disappeared due to an absence of food, but many others declined as the structural complexity of the habitat decreased due to erosion of the dead coral substratum. There are also studies showing strong correlation between structural complexity and species recruitment, abundance, or diversity in ecosystems other than coral reefs (e.g., Hicks 1980; Connell and Jones 1991; Fernandez et al. 1993; Carr 1994; Herrnkind and Butler 1994; Szedlmayer and Able 1996).

Structural complexity provides smaller species with living space, increased food abundance, and refuge from predation (Sebens 1991). For example, Bros (1987) found that species diversity increased when artificial barnacle shells were added to a smooth surface. Presumably the increased surface area and presence of small spaces provided habitat for additional species. Lowered vulnerability to predators is another important aspect of habitat structural complexity. Prey abundance was greater in seagrass beds (Nelson 1979), worm tube aggregations (Woodin 1978), mussel clumps (Witman 1985), and algal turfs (Coull and Wells 1983; Marinelli and Coull 1987) than in less-structured bottoms when predators were present. It is now almost axiomatic that the more diverse marine habitats have higher species diversity (Sebens 1991).

The reasons why structural complexity is essential for many benthic species become clearer upon examining the relationship of organisms to the fluid dynamics just above the seabed. Most of the world's seabed consists of unconsolidated, fine, muddy sedi-

ments, where the sediment–water interface would be essentially flat but for the living things that increase structural complexity both above and below the sediment surface. Many seabed organisms are suspension feeders, orienting themselves with currents that bear food particles. Drag sharply decreases current velocities in the few centimeters above the sediment, decreasing opportunities for suspension-feeding (see reviews by Butman 1987; Snelgrove and Butman 1994). As a result, benthic organisms that raise their feeding structures even one or a few centimeters into the water column are better situated to capture plankton and detritus carried by currents. Furthermore, because the oxygen content of seawater is more than four orders of magnitude lower than that of air, respiration in and on the seabed rapidly depletes oxygen in the millimeter-to-centimeter-thick bottom boundary layer that sits just above the seabed (Jørgensen 1996), with the result that, on mud bottoms, sediment 1 cm or more below the sediment–water interface is almost always devoid of oxygen (see review by Watling 1991). Because anoxia is inimical to nearly all benthic animals, many infauna that make tunnels or tubes within the seabed generate currents that break through the bottom boundary layer, bringing the infauna oxygenated water and food particles. Other infauna and many epibiota avoid the oxygen-poor conditions of the sediment by placing their respiratory structures above the bottom boundary layer.

The structures that benthic species create increase seabed structural complexity. Many other species, including species sought as food by fishes, that do not colonize soft substrata per se live on or in these biogenic structures. For these reasons, structures—even small ones—are more important for epibiota on the seabed than on the land. Hard surfaces in the sea are generally far more densely colonized than hard surfaces on land, including rainforests, with their abundance of epiphytes and associated animals (E. A. Norse, personal observations). The diversity of benthic infauna and epibiota, therefore, provides essential habitat features including structures and food that sustain many of the world's commercial fishes (Boehlert 1996).

Changes in Species Composition at Reduced Structural Complexity

In terrestrial ecosystems, species composition is determined largely by the spatial configuration of structure-forming species; ancient coniferous forests,

tallgrass prairies, and sandy deserts have very different assemblages of species. Structures that are essential to some species are unnecessary or even disadvantageous to others; removing structures frees up resources for species that do not need structures. For example, a Pacific Northwest wildfire or logging operation that eliminates ancient western red cedars *Thuja plicata* and northern flying squirrels *Glaucomys sabrinus* creates opportunities for fireweeds *Epilobium angustifolium* and creeping voles *Microtus oregoni*. Species composition is so closely tied to structure that terrestrial wildlife biologists have long manipulated habitat structure to maximize populations of species they consider desirable, such as deer *Odocoileus* spp.

Because fishery biologists have (until very recently) been less attuned to effects of small seabed structures, habitat relationships of fishes, especially postsettlement stages, are far less known in the sea and have largely been overlooked in fishery management. However, ecological theory and ubiquitous observations both suggest that severe disturbances that remove structure from the seabed will profoundly change species composition, harming many species but favoring some others, thereby decreasing species diversity. In this regard, trawling and dredging have effects similar to organic enrichment, which reduces species diversity and produces communities comprised of large numbers of a few opportunistic species (Pearson and Rosenberg 1978). A small but growing body of studies from places where scientists have looked at effects of mobile fishing gear, including Northern Europe, Australia, New Zealand, and the Atlantic and Pacific coasts of North America, support this hypothesis.

In the North Sea, where all the large *Sabellaria spinulosa* polychaete reefs were deliberately removed, species typical of open sands now dominate and support significant flatfish fisheries (Riesen and Reise 1982). In Loch Gareloch on the Irish Sea, trawling significantly reduced populations of some infauna (e.g., the nut clam *Nucula nitidosa*), while opportunistic cirratulid and capitellid polychaetes became more abundant (Lindeboom and de Groot 1998). In northwestern Australia, Sainsbury (1987, 1988) found high-value *Lethrinus* (emperors), *Lutjanus* (snappers or seaperch), and *Epinephelus* (groupers or rockcod) dropped from 45 to 77% of the catch to 15% after trawling removed structure-forming sponges and gorgonians. At the same time, commercially less-valued species characteristic of sandy bottoms in the genera *Nemipterus* (threadfin-

bream) and *Saurida* (lizardfishes or grinners) became more abundant. In Hauraki Gulf, New Zealand (North Island), Thrush et al. (1998) found that areas with the least disturbance from trawling, seining, and scallop dredging had the most long-lived surface-dwelling invertebrates, the smallest proportion of opportunistic species, and the highest species diversity (using one kind of sampling gear) and highest density of large individuals and most organisms (using another type of gear). On Georges Bank off New England, Collie (1998) reported that mobile fishing gear on gravel bottoms removed the three-dimensional cover provided by epifauna, with undisturbed areas having higher abundance, biomass, and species diversity as indicated by the presence of fragile species such as sponges, nudibranchs, worms, and small fishes, while areas subjected to bottom trawling and scallop dredging were characterized by scavengers such as hermit crabs and sea stars. Finally, off the Big Sur coast of California, Engel and Kvitek (1998) found that heavily trawled areas have a low diversity of polychaete worms but large populations of an opportunistic amphinomid polychaete *Chloeia pinnata*, which the authors found to be the dominant prey item of several flatfish species. In these cases, trawling tended to eliminate competitively dominant, long-lived but disturbance-sensitive structure-forming benthic species, freeing up food and space for shorter-lived, disturbance-insensitive, opportunistic (weedy) species. In the absence of needed benthic structures or foods, groupers and cod disappear but lizardfishes and flatfishes fare better. Trawling and dredging decrease species diversity but increase populations of disturbance-tolerant benthic species and fishes that eat them, just as clear-cutting eliminates ancient forests and spotted owls and shifts production toward grasses and grazers.

Mobile Fishing Gear Effects, Type Conversion, and Sustainability

Bottom trawls and dredges used to catch benthic and demersal fishes, crabs, lobsters, shrimps, bivalves, sea urchins, and corals disturb the seabed in ways that overturn rocks, flatten sand waves, and crush, bury, and expose benthic organisms and biogenic structures (see reviews by Auster and Langton 1999 and Watling and Norse 1998). In the past, sizable structures (e.g., boulders) prevented trawling, but the advent of rockhopper and streetsweeper gear now allows trawling on virtually any kind of bottom, and fish finders and global positioning systems

allow fishers to locate good spots and relocate them accurately until the spots are no longer so good. Moreover, the progressive disappearance of high-value commercial fishes in shallow waters has pushed fishing ever deeper; Merrett and Haedrich (1997) noted that trawling occurs as deep as 2,000 m, covering a total area of approximately 2.5 km^2 during each tow. Trawlers are more powerful than in the past, and improved technologies allow trawlers to fish deeper, farther offshore, and on rougher bottoms (Mirarchi 1998). The technological and economic forces that have increased fishing power and intensity have brought unprecedented disturbance to the seabed worldwide. Ecosystems with high structural complexity are likely to change most as fishing pressure increases (Auster 1998).

The use of mobile fishing gear is now the most important source of anthropogenic disturbance of the seabed and the principal agent of disturbance (anthropogenic or natural) in deep shelf, slope, and seamount waters where disturbance frequencies are naturally low. Watling and Norse (1998) have now shown that trawling occurs on a scale that had not previously been imagined; worldwide, an area equaling about half of the continental shelf—an area twice as large as the lower 48 U.S. states combined—is trawled every year. The few specific areas for which data are available are trawled at return intervals (average time between successive disturbances) ranging from years down to months.

In gauging the impact of a disturbance, it is useful to compare its return time with the time required for succession to restore the ecosystem's original structure. Impacts are more worrisome as return intervals become a significant fraction of the time until successional climax, because these return intervals shift the successional mosaic toward one dominated by recently disturbed patches. In many forest communities, biologists know the time needed for communities to attain late-successional characteristics. Much less is known about succession in many continental shelf, slope, and seamount areas, but a very crude estimate can come from knowing the life span of key structure-forming species. This assumes that these structure-forming species can colonize recently disturbed patches; alternatively, they could require intermediate successional stages before becoming established. Pacific Northwest Douglas-fir and western hemlock communities start to develop late-successional (ancient forest) attributes at about 200 years, and the dominant structure-formers have maximum life spans of 500–1,200

years, so disturbance return times (logging rotations) of anything less than 200 years essentially eliminate late-successional forests from the landscape matrix.

Life spans of marine structure-forming species are less known than they are for forest trees, but they range from months or years to several centuries (maximum estimated longevity for ocean quahog clams *Arctica islandica* is 221 years [Kraus et al. 1989]) or even more (gorgonian corals in the genus *Primnoa* can reach 500 or perhaps even 1,500 years in age [Risk et al. 1998]). It is reasonable to assume that recovery times in benthic ecosystems range from months to millennia, typically (on the continental shelf) ranging from years to decades. Because disturbance return times are short in comparison—for example, four months on Georges Bank (Auster et al. 1996), one year in the Gulf of Maine (Auster et al. 1996), and a worldwide continental shelf average of roughly two years (Watling and Norse 1998)—mobile fishing gear often disturbs the seabed much faster than succession and other benthic processes can restore seabed structure, converting ecosystems dominated by structure-forming and structure-needing species to ecosystems dominated by other species. The terrestrial equivalent of this would be wholesale, worldwide, unplanned, and unchronicled conversion of virgin forest to cattle pasture.

In the Irish Sea, where trawling has occurred intensively, the IMPACT-II report (Lindeboom and de Groot 1998:361) stated, "The present species-poor and low biomass fauna may represent an artificial man-made community adapted to the regular fishing disturbance experienced at this site" and concluded (p. 364), "if trawling intensity remains high, these communities may never recover." Foresters call this kind of anthropogenic change "type conversion," a practice prohibited except in extraordinary circumstances under the U.S. National Forest Management Act, the federal law that governs extraction and replacement of trees on most federally owned multiple-use forestlands. Strangely enough, the Magnuson-Stevens Fishery Conservation and Management Act of 1996 (also called the Sustainable Fisheries Act) does not even address ecosystem conversion, despite the fact that mobile fishing gears are converting structurally diverse benthic ecosystems to essentially featureless plains at a rate two orders of magnitude faster than forests are being converted worldwide. Trawling and dredging could be one of the least-known factors affecting the world's biological diversity.

It has become clear in this decade that marine biodiversity is increasingly threatened (Norse 1993; Butman and Carlton 1995). At the same time, many of the world's demersal fisheries have shown alarming downward trends (FAO 1997). Although it is clear that many fish species are being caught at rates their populations cannot sustain, it is no less clear that demersal fish habitat is being stripped of its essential structural complexity. Which of these two contributing factors is more important is not yet known (Fogarty and Murawski 1998), but it is the height of folly to think that overexploitation is the only way that fishing decreases fisheries yields. It is also apparent that areas supporting some demersal fisheries, including brown shrimp *Penaeus aztecus* in the northern Gulf of Mexico and plaice *Pleuronectes platessa* in the North Sea, have been trawled for many years without marked decreases in catch after their initial conversion. These may be canonical examples of fisheries based on opportunistic, disturbance-tolerant species.

In view of the profound effects of mobile fishing gear on benthic ecosystems, it is remarkable that there is no management structure in place in the United States (or anywhere else that we know about) charged with maintaining the seabed's biological diversity. As Boehlert (1996:33) noted, "legal authority under the Magnuson Fishery Conservation and Management Act (under which fishery management plans are developed) gives no consideration to genetic, species, or ecosystem biodiversity except as it affects protected species or critical habitats." This situation remained unchanged when the Magnuson Fishery Conservation and Management Act was reauthorized as the Sustainable Fisheries Act of 1996; only habitat essential to the well-being of fishes is given consideration. Areas of the seabed where fish are likely to roam, but are not known to be essential to any life history stage, are outside the management requirements of the Sustainable Fisheries Act. Consequently, there are no provisions to limit habitat destruction and biodiversity loss anywhere that is not designated as essential fish habitat. Nobody is safeguarding the seabed from fishing.

In January 1998, 1,605 marine scientists and conservation biologists from 70 nations issued a statement called "Troubled Waters: A Call for Action" (MCBI 1998). The statement called upon citizens and governments worldwide to "Ameliorate or stop fishing methods that undermine sustainability by harming the habitats of economically valuable marine species and the species they use for food and shelter." The question that fishery biologists, fishers, conservationists, managers, legislators, and the public must ask is whether we are willing to live in

a world where spotted owls, cod, and groupers become as vanishingly rare as their rugose habitats, to be superceded by cattle, plaice, and lizardfishes. To people concerned only about the gross tonnage of meat produced, such questions might not be troubling; there are almost always some organisms opportunistic enough to survive even where disturbance is severe and chronic. But to a growing number of people, including thousands of leading scientists, the loss of marine biodiversity is an appalling prospect.

Any alternative to the current approach must take legislative and management steps to both protect substantial areas of seabed from becoming structurally simplified and to restore the seabed's structure, species composition, and functioning. The actions we take ultimately hinge on whether we value the living sea as anything more than a wet, salty cattle ranch.

The difficult task of balancing short-term economic gains with maintenance of biodiversity and longer-term economic benefits involves recognizing economic behaviors of people who take wild living resources. Loggers prefer large, high-quality, high-value trees and focus their attention in forest areas having them. As Norse (1990) noted, U.S. National Parks and Wilderness Areas tend to be located in areas of low biological and economic productivity, such as scenic, craggy snow-covered mountains that lack trees sought by loggers; these areas have a low diversity of forest species. In a similar way, fishers concentrate trawling and dredging effort in certain areas (see Figure 1 in Mirarchi 1998). Some other areas (quite likely areas with the lowest habitat value for fishes) escape disturbance from fishing. However, protecting areas that nobody wants because they are biologically unproductive does little to maintain biodiversity.

Lessons Learned

Marine conservation lags behind terrestrial conservation, both in terms of what scientists know and in the creation and implementation of laws to protect resources. Lawmakers and marine fisheries managers are only now awakening to something their terrestrial counterparts have known for two decades: that human-caused disturbance is dramatically reducing biological diversity, and that to avoid undesirable losses, disturbance frequency or severity must be reduced. Although the seabed is a crucial component of the Earth's biological diversity, the prevailing marine fisheries paradigm focuses on managing populations in isolation from their environment. This paradigm has pushed populations of

many high-value fishes so far below maximum sustainable populations that the world's fish catch is increasingly comprised of low-trophic-level "baitfishes" rather than higher-trophic-level fishes (Pauly et al. 1998). Foot-dragging and "more-of-the-same" fishery legislation, management, and scientific research are a guaranteed recipe for further losses, not only of the commercial fisheries that are the focus of U.S. laws, but, more broadly, of the biological diversity that supports fisheries.

In the nearly two decades since biological diversity loss was defined as the world's premier conservation challenge, scientific and managerial advances have strengthened conservation in the terrestrial and freshwater realms. Yet the United States has no federal laws focused on maintaining biological diversity in the sea and nothing remotely approaching the multidisciplinary analysis and decision making that led to conservation of spotted owls and their Pacific Northwest ancient forest habitat. The biodiversity ethic that has become the driving force in nonmarine conservation has yet to make substantial inroads in the marine realm; marine conservation is still largely about maximizing the fish catch or preventing a few other preferred species (especially marine mammals) from harm. As the sea loses biological diversity at an accelerating rate, it is clear that a different approach is needed.

Although some die-hards will undoubtedly deny the importance of trawling and scallop dredging impacts no matter how strong the evidence is, there are fishermen—at least when they are speaking anonymously—who know what marine scientists have only recently learned, as the following quotes from Nova Scotia and New Brunswick fishermen reveal (Fuller and Cameron 1998):

"Draggers have leveled off Western Bank. During the '70s and '80s they tore all the plant life off it. This has the same effect as clearcutting." (Respondent 1)

"There used to be an awful mess of [tree-like corals] and the nets got tore to pieces. We got them pretty much cleaned up. We used to clean out the trees when hauling back the nets." (Respondent 14)

"There shouldn't be dragging, it tears the plant life off the bottom. It might take ten years to come back. You can't take a plow through a field and expect the grass to grow back right away." (Respondent 29)

"Rockhopper gear changes the bottom and gets rid of places fish can hide." And "Now they scallop 24 hours a day, all winter long. There are more boats and more power to tow with, this causes the gear to dig in better ... They drag up everything and it doesn't

have a chance to come back." And "If the system were left alone, it will recover somewhat. It needs time to heal." (Respondent 3)

We believe that the United States and other nations need to make all human activities in the sea—whether shipping, oil and gas production, recreation, or fishing—compatible with maintaining and restoring biological diversity. As on land, we need intelligent, flexible, scientifically sound, and carefully monitored limits on our take of marine wildlife as well as a comprehensive system of protected areas that are managed to maintain marine biodiversity. In practical terms, that means that a substantial portion of the sea (the signers of "Troubled Waters" called for 20%) must be off-limits to any activity, including trawling and dredging, that significantly reduces biological diversity. The essential fish habitat provisions in the 1996 Magnuson-Stevens Fishery Conservation and Management Act are a step in the right direction, but unless the provisions are strengthened to address broader biodiversity needs, they are not sufficient. There need to be zones in the sea where people can fish and other zones where the marine life can recruit, grow, and spawn free from fishing pressure, just as wildlife can in terrestrial national parks.

Acknowledgments

References

Ausich, W. I., and D. J. Bottjer. 1982. Tiering in suspension-feeding communities on soft substrata throughout the Phanerozoic. Science 216:173–174.

Auster, P. J. 1998. A conceptual model of the impacts of fishing gear on the integrity of fish habitat. Conservation Biology 12:1198–1203.

Auster, P. J., and eight coauthors. 1996. The impacts of mobile fishing gear on seafloor habitats in the Gulf of Maine (Northwest Atlantic): implications for conservation of fish populations. Reviews in Fisheries Science 4:185–202.

Auster, P. J., and R. W. Langton. 1999. The effects of fishing on fish habitat. Pages 150–187 in L. R. Benaka, editor. Fish habitat: essential fish habitat and rehabilitation. American Fisheries Society, Symposium 22, Bethesda, Maryland.

Bell, J. D., and R. Galzin. 1984. Influence of live coral cover on coral-reef fish communities. Marine Ecology Progress Series 15:265–274.

Boehlert, G. W. 1996. Biodiversity and the sustainability of marine fisheries. Oceanography 9:28–35.

Bohnsack, J. A. 1991. Habitat structure and the design of artificial reefs. Pages 412–426 in S. S. Bell, E. D. McCoy and H. R. Mushinsky, editors. Habitat structure: the arrangement of objects in space. Chapman and Hall, New York City.

Bros, W. E. 1987. Effects of removing or adding structure (barnacle shells) on recruitment to a fouling community in Tampa Bay, Florida. Journal of Experimental Marine Biology and Ecology 105:275–296.

Butman, C. A. 1987. Larval settlement of soft-sediment invertebrates: the spatial scales of pattern explained by active habitat selection and the emerging role of hydrodynamical processes. Oceanography and Marine Biology Annual Review 25:113–165.

Butman, C. A., and J. T. Carlton, editors. 1995. Understanding marine biodiversity: a research agenda for the nation. National Academy Press, Washington, D.C.

Carr, M. H. 1994. Effects of macroalgal dynamics on recruitment of a temperate reef fish. Ecology 75:1320–1333.

Collie, J. 1998. Studies in New England of fishing gear impacts on the sea floor. Pages 53–62 in E. M. Dorsey and J. Pederson, editors. Effects of fishing gear on the sea floor of New England. Conservation Law Foundation, Boston.

Connell, S. D., and G. P. Jones. 1991. The influence of habitat complexity on postrecruitment processes in a temperate reef fish population. Journal of Experimental Marine Biology and Ecology 151:271–294.

Coull, B. C., and J. B. J. Wells. 1983. Refuges from fish predation: experiments with phytal meiofauna from the New Zealand rocky intertidal. Ecology 64:1599–1609.

Diamond, J. M. 1975. The island dilemma: lessons of modern biogeographic studies for the design of natural preserves. Biological Conservation 7:129–146.

Ebeling, A. W., and M. A. Hixon. 1991. Tropical and temperate reef fishes: comparison of community structures. Pages 509–563 in P. F. Sale, editor. The ecology of fishes on coral reefs. Academic Press, San Diego.

Engel, J., and R. Kvitek. 1998. Impacts of otter trawling on a benthic community in Monterey Bay National Marine Sanctuary. Conservation Biology 12:1204–1214.

FAO (Food, and Agriculture Organization of the United Nations). 1997. Review of the state of the world fishery resources: marine fisheries. Fisheries circular number 920 FIRM/C920. FAO, Rome.

Fernandez, M., O. Iribarne, and D. Armstrong. 1993. Habitat selection by young-of-the-year Dungeness crab Cancer magister Dana and predation risk in intertidal habitats. Marine Ecology Progress Series 92:171–177.

Fogarty, M. J., and S. A. Murawski. 1998. Large-scale disturbance and the structure of marine systems: fishery impacts on Georges Bank. Ecological Applications 8(1)Supplement:S6–S22.

Fuller, S., and P. Cameron. 1998. Marine benthic seascapes: fishermen's perspectives. Marine Issues Committee Special Publication 3, Ecology Action Centre, Halifax, Nova Scotia.

Gage, J. D., and P. A. Tyler. 1991. Deep-sea biology, a natural history of organisms at the deep-sea floor. Cambridge University Press, New York.

Giere, O. 1993. Meiobenthology, the microscopic fauna in aquatic sediments. Springer-Verlag, New York.

Gotceitas, V., and J. A. Brown. 1993. Substrate selection by juvenile Atlantic cod (Gadus morhua): effects of predation risk. Oecologia 93:31–37.

Hansen, A. J., W. C. McComb, R. Vega, M. G. Raphael, and M. Hunter. 1995. Bird habitat relationships in natural and managed forests in the west Cascades of Oregon. Ecological Applications 5:555–569.

Herrnkind, W. F., and M. J. Butler IV. 1994. Settlement of spiny lobster, Panulirus argus (Latreille, 1804), in Florida: pattern without predictability? Crustaceana 67:46–64.

Heywood, V. H., editor. 1995. Global biodiversity assessment. Cambridge University Press, New York.

Hicks, G. R. F. 1980. Structure of phytal harpacticoid copepod assemblages and the influence of habitat complexity and turbidity. Journal of Experimental Marine Biology and Ecology 44:157–192.

Jørgensen, B. B. 1996. Material flux in the sediment. Pages 115–135 in B. B. Jorgensen and K. Richardson, editors. Eutrophication in the marine environment. Coastal and estuarine studies, volume 52. American Geophysical Union, Washington, D.C.

Kraus, M. G., B. F. Beal, and S. R. Chapman. 1989. Growth rate of Arctica islandica Linne: a comparison of wild and laboratory-reared individuals. Journal of Shellfisheries Research 8:463.

Lindeboom, H. J., and S. J. de Groot, editors. 1998. IMPACT-II. The effects of different types of fisheries on North Sea and Irish Sea benthic ecosystems. Netherlands Institute for Sea Research (NIOZ), Texel, Netherlands.

Lovejoy, T. 1980. A projection of species extinctions. Pages 328–332 in Council on Environmental Quality and U.S. Department of State. The global 2000 report to the President, volume 2, the technical report. U.S. Government Printing Office, Washington, D.C.

MacArthur, R. H., and J. W. MacArthur. 1961. On bird species diversity. Ecology 42:594–598.

Marinelli, R. L., and B. C. Coull. 1987. Structural complexity and juvenile fish predation on meiobenthos: an experimental approach. Journal of Experimental Marine Biology and Ecology 108:67–81.

McAllister, D. E. 1991. What is the status of the world's coral reef fishes? Sea Wind 5:14–18.

MCBI (Marine Conservation Biology Institute). 1998. Troubled waters: a call for action. http://www.mcbi.org/trouble1.html (accessed 3 November 1998).

Merrett, N. R., and R. L. Haedrich. 1997. Deep-sea demersal fish and fisheries. Chapman and Hall, New York.

Mirarchi, F. 1998. Bottom trawling on soft substrates. Pages 80–84 in E. M. Dorsey and J. Pederson, editors. Effects of fishing gear on the sea floor of New England. Conservation Law Foundation, Boston.

Myers, N. 1979. The sinking ark: a new look at the problem of disappearing species. Pergamon, New York City.

Nelson, W. G. 1979. Experimental studies of selective predation on amphipods: consequences for amphipod distribution and abundance. Journal of Experimental Marine Biology and Ecology 38:225–245.

Norse, E. A. 1990. Ancient forests of the Pacific Northwest. Island Press, Washington D.C.

Norse, E. A., editor. 1993. Global marine biological diversity: a strategy for building conservation into decision making. Island Press, Washington, D.C.

Norse, E. A., and R. E. McManus. 1980. Ecology and living resources: biological diversity. Pages 31–80 in Environmental quality 1980. Council on Environmental Quality, Washington D.C.

Ormond, R. F. G., and C. M. Roberts. 1997. The biodiversity of coral reef fishes. Pages 216–257 *in* R. F. G. Ormond, J. D. Gage, and M. V. Angel, editors. Marine biodiversity: patterns and processes. Cambridge University Press, Cambridge UK.

Pauly, D., V. Christensen, J. Dalsgaard, R. Froese, and F. Torres Jr. 1998. Fishing down marine food webs. Science 279:860–863.

Pearson, T. H., and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology Annual Review 16:229–311.

Perry, D. A. 1994. Forest ecosystems. Johns Hopkins University Press, Baltimore.

Reaka-Kudla, M. 1997. The global biodiversity of coral reefs: a comparison with rain forests. Pages 83–108 *in* M. L. Reaka-Kudla, D. E. Wilson and E. O. Wilson, editors. Biodiversity II: understanding and protecting our biological resources. Joseph Henry Press, Washington, D.C.

Riesen, W., and K. Reise. 1982. Macrobenthos of the subtidal Wadden Sea: revisited after 55 years. Helgolander Meeresuntersuchungen 35:409–423.

Risk, M. J. 1972. Fish diversity on a coral reef in the Virgin Islands. Atoll Research Bulletin 153:1–6.

Risk, M. J., D. E. McAllister, and L. Behnken. 1998. Conservation of cold- and warm-water seafans: threatened ancient gorgonian groves. Sea Wind 12(1):2–21.

Sainsbury, K. J. 1987. Assessment and management of the demersal fishery on the continental shelf of northwestern Australia. Pages 465–503 *in* J. J. Polovina and S. Ralston, editors. Tropical snappers and groupers: biology and fisheries management. Westview Press, Boulder, Colorado.

Sainsbury, K. J. 1988. The ecological basis of multispecies fisheries and management of a demersal fishery in tropical Australia. Pages 349–382 *in* J. A. Gulland, editor. Fish population dynamics, 2nd edition. Wiley, New York.

Sano, M., M. Shimizu, and Y. Nose. 1984. Changes in structure of coral reef fish communities by destruction of hermatypic corals: observational and experimental views. Pacific Science 38:51–79.

Sebens, K. P. 1991. Habitat structure and community dynamics in marine benthic systems. Pages 199–234 *in* S. S. Bell, E. D. McCoy, and H. R. Mushinsky, editors. Habitat structure: the arrangement of objects in space. Chapman and Hall, New York.

Sharp, G. D. 1988. Fish populations and fisheries: their perturbations, natural and man-induced. Pages 155–202 *in* H. Postma and J. J. Zijlstra, editors. Continental shelves. Ecosystems of the world, volume 27. Elsevier Publishers, Amsterdam.

Snelgrove, P. A., and C. A. Butman. 1994. Animal-sediment relationships revisited: cause versus effect. Oceanography and Marine Biology Annual Review 32:111–177.

Szedlmayer, S. T., and K. W. Able. 1996. Patterns of seasonal availability and habitat use by fishes and decapod crustaceans in a southern New Jersey estuary. Estuaries 19:697–709.

Terborgh, J. 1974. Preservation of natural diversity: the problem of extinction-prone species. BioScience 24:715–722.

Thrush, S. F., and nine coauthors. 1998. Disturbance of the marine benthic habitat by commercial fishing: impacts at the scale of the fishery. Ecological Applications 8:866–879.

Watling, L. 1991. The sedimentary milieu and its consequences to resident organisms. American Zoologist 31:789–796.

Watling, L., and E. Norse. 1998. Disturbance of the seabed by mobile fishing gear: a comparison to forest clearcutting. Conservation Biology 12:1180–1197.

Wheatcroft, R. A., P. A. Jumars, C. R. Smith, and A. R. M. Nowell. 1990. A mechanistic view of the particulate biodiffusion coefficient: step lengths, rest periods, and transport directions. Journal of Marine Research 48:177–207.

Witman, J. D. 1985. Refuges, biological disturbance, and rocky subtidal community structure in New England. Ecological Monographs 55:421–455.

Woodin, S. A. 1978. Refuges, disturbance, and community structure: a marine soft-bottom example. Ecology 59:274–284.