

3

Ecological Comparison

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Introduction

The ecology of fish resources is part of a larger picture defined as the fish chain. Diversity is at the base of most ecological processes involving resources and its alteration is viewed as a major source of large ecological and societal changes (Chapin et al. 2000). In addition, diversity is easy to define and conceptualise and is probably the best-studied ecological variable. Differences in the diversity of exploited species are extremely important, for example, the approximately thirty species commonly exploited in the Northeast Atlantic as compared with well over two hundred species in the tropical Western Pacific. Consequences may be numerous at any level of the fish chain, as is illustrated in fig. 3.1.

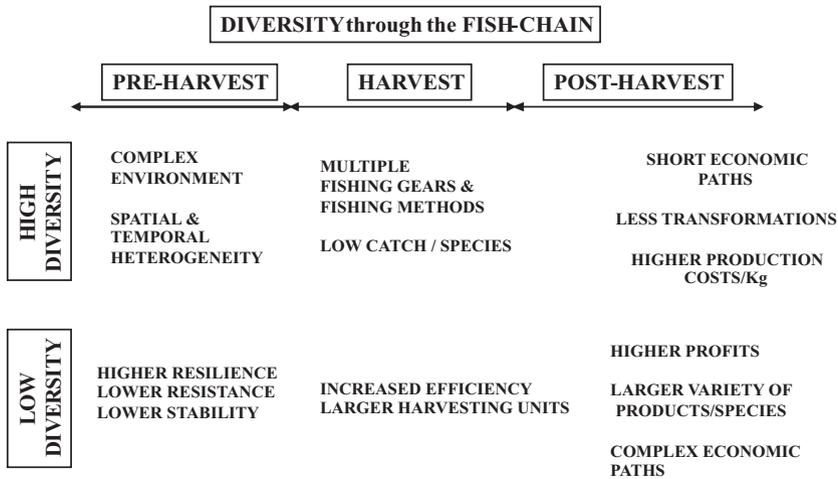


Fig. 3.1 Possible consequences of diversity for fish chains

Diversity has many meanings in marine ecology (Steele 1991). It can apply to a continuum of organisation levels ranging from genetic diversity to ecosystem and landscape diversity. All these levels may be viewed as linked and the factors affecting one level of diversity usually affect the other ones as well (fig. 3.2).

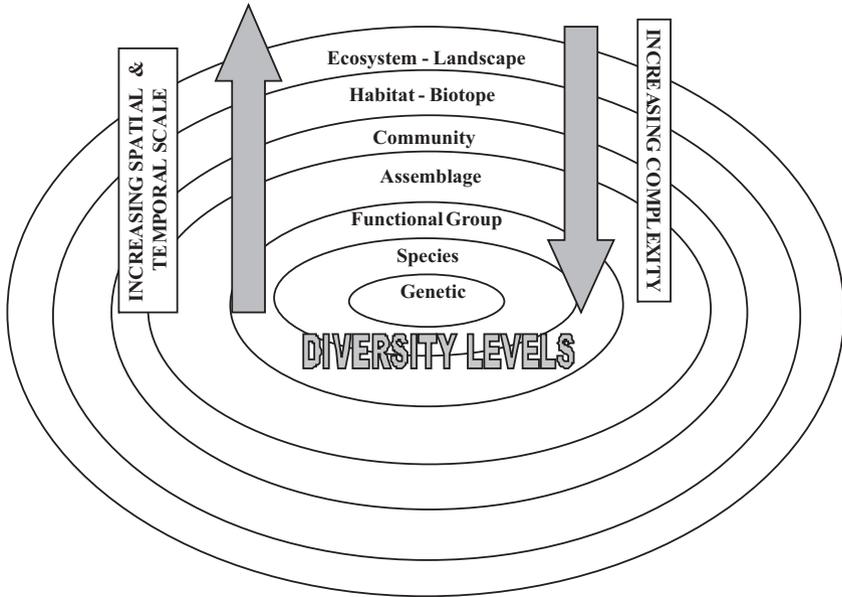


Fig. 3.2 Diversity with increasing levels of organisation

The ascending arrow indicates that, moving from the genetic to the ecosystem level, changes in diversity depend on larger spatial scales and longer time scales. The descending arrow shows that the total number of components increases as the organisation level decreases.

Diversity can be essentially approached from three perspectives. The first is composition; e.g., of the species or functional groups in an assemblage. The second is the relative abundance of species in an area, as is usually expressed by indices related to the evenness of the distribution of individuals among the total number of species (Legendre and Legendre 1998). The indices make it possible to compare populations or assemblages through time for the same or different areas. The third perspective is the number of items at any organisational level (fig.3.2), e.g., the number of species in an assemblage. This is often referred to as richness and can be considered at various spatial scales. In this chapter we are interested in the following:

- *Regional diversity*: the number of species known in a region. At the taxonomic level, it usually refers to a species checklist, but can also be considered at the functional group, genetic, or other levels.
- *Island or province diversity*: the number of taxa known for a sub-region, often called a province in ecology, or an island.
- *Local diversity*: the number of taxa (or functional groups or genetic varieties) in a specific habitat, e.g., the number of fish species at a specific reef, mangrove area or trawling ground.
- *Species density or richness*: the number of taxa in each standard sampling unit, e.g., transect or trawl.

Regional diversity is at the base of the concept of a regional meta-community, i.e., the pool of species at the regional level. From this pool, island or province, meta-communities are derived, which are the species pools the species assemblages are drawn from in a specific habitat, e.g., the fish assemblage at a particular reef.

Factors and Scales Affecting Diversity

Types of Factors and Scale Range

In all three of its meanings, diversity is affected by a range of factors at all the spatial and temporal scales. It might be convenient to split the factors into large-scale and local-scale groups and recognise a continuum between them. Large-scale spatial factors may be linked to physical phenomena such as regional upwellings, island size, island type, connectivity between islands, or regions or evolutionary phenomena such as biodiversity centres of origin or dispersal routes. These factors are not linked to human influence. Local-scale spatial factors include physical factors (e.g., depth, coastal configuration, and terrestrial input) as well as ecological factors such as biotope type (e.g., mangrove, reef, soft bottom), habitat (e.g., reef flat, reef slope, and back reef), components of the habitat (e.g., mangrove height, coral cover, and sea-grass density), and human factors such as fishing level and pollution. With the exception of some factors such as depth, most local factors are susceptible to human influence.

Time scales usually refer to perturbations. Short-term perturbations may involve cyclical changes (e.g., seasons) as well as acute pollution, catastrophic climatic events such as storms or floods, coral bleaching and temperature disruption. Mid-term perturbations cover events that are usually less intense but have a longer duration such as fishing, chronic pollution, invasive species, climatic events such as El Niño and their consequences (droughts, temperature, and salinity changes). Long-term perturbations are less easy to perceive and represent events such as sea level rise, long-term temperature shifts, and their consequences (e.g., current patterns), changes in land use (e.g., deforestation of the Amazon basin, construction of major dams, long-term effects of fishing). Even though the role of humans is not always clearly established, there is usually some anthropogenic influence in most time-related factors acting on diversity.

Factors and Scale Interaction

It is important to note that spatial and temporal factors interact. In particular, large-scale spatial factors are mainly affected by long-term perturbations and all the time-scale levels influence local-scale factors. All these factors essentially have two effects on diversity. Firstly, they induce perturbations

so that composition, richness, and evenness may be affected at all the organisational levels. Secondly, these factors structure diversity, with the changes depending on the factors and their intensity.

There is a relation between the scale where factors intervene and the organisational level where they act (Sale and Guy 1992). In general, the higher the organisational level, the larger the spatial and time scales (fig. 3.2, see also Hatcher 1997 for coral reefs). To understand the variations observed at the level of a single specimen, the scale is limited to the immediate environment of this fish and its life span. At higher organisational levels, the spatial and time scales necessary to comprehend changes become larger. The major problem facing the ecologist is that the larger the scale, the less information is available. The paradox, however, is that at the higher scales the crucial factors are easier to detect and measure. If we are interested in reef fish, measuring the effects of local factors such as coral cover, habitat complexity, perturbations due to fishing, pollution, and so forth may be very complex. Conversely, when considering the regional scale, the major factors are island size, island type (high or low island, opening of the lagoon influence), the connectivity between islands (function of the distance and size of nearby islands) and so forth, factors that are easy to measure and can be rather simply integrated into models.

Diversity at Various Scales

Species have particular habitat needs. This means that on a local scale, species are found under specific conditions. A basic law of ecology states that there is a strong relation between the number of habitats and the total number of species in an area. Furthermore, species diversity tends to increase with habitat complexity or heterogeneity. Habitat complexity can be scaled for the major coastal marine biotopes (fig.3.3). Many factors act on habitat complexity or heterogeneity. Firstly, there are regional factors such as latitude, biogeographical region, regional climate, large-scale geomorphology, and geology or island size. In comparing North-South situations, habitats are usually more complex in warm climates, e.g., more complex in the Indo-Pacific than in the Atlantic. This large-scale component is often overlooked when examining fisheries management, since most models do not take into account such regional factors even though they can play major roles in diversity and consequently in resource levels. There are also local factors regarding habitat heterogeneity, in particular depth, salinity, and temperature as well as perturbations, especially fishing and pollution.

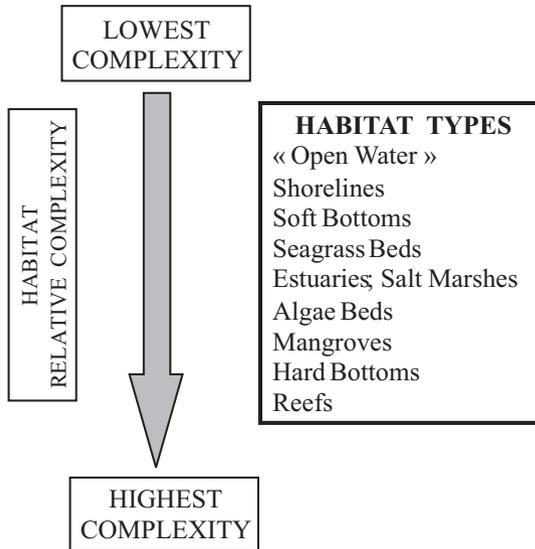


Fig. 3.3 Classification of some major coastal marine habitats according to complexity

Humans have little power to increase habitat diversity (number of habitats in an area). The method most frequently used to try to increase marine habitat diversity is the creation of artificial reefs. This is often, though not always, a minor change compared to the huge adverse impacts on habitat diversity caused by coastal works, pollution, certain types of fishing (in particular trawling), the introduction of alien species and so forth. However, humans can restore habitat complexity and heterogeneity to some extent by limiting the perturbation levels of these factors.

Why do we think diversity is so important to the fish chain? After all, fishers do not sell species, they sell fish and usually the more fish they sell, the better (though this is debatable as fish size and quality come into play). The fact is that there is a strong relation between diversity and fish density or fish biomass. This type of relation has a theoretical background (Hubbell 2001) indicating that the higher the number of species in an area, the larger the number of species with large densities or large biomasses.

Diversity and Ecological Functioning

Ecosystem functioning is based mainly on the variability of diversity in the broadest sense of the word. In particular, diversity is instrumental in three important characteristics of ecosystems: stability, resistance and resilience (McCann 2000). An ecosystem is stable if it does not deviate from an average state. An ecosystem is resistant if it requires a great deal of perturbation to make it deviate from this stable state. Resilience is the ease with which an ecosystem returns to its former stable state. It is important to note that a

stable state does not necessarily mean equilibrium, since ecosystems are admittedly never in equilibrium for long but perpetually change from one state to another, depending on their environment.

If a fish assemblage is submitted to a perturbation, it tends to change. Once the conditions return to their initial state, the assemblage should as well. Ecological theory formulates the hypothesis that systems with high diversity have more resistance but less resilience than systems with fewer species (McCann 2000). In other words, fish assemblages such as those on reefs with very high numbers of species tend to resist perturbations for a long time but once they start changing, a return to their initial state is slow and may be impossible. Conversely, simple fish assemblages such as those on soft bottoms are less resistant to perturbations but more apt to return quickly to their initial state once the disturbance is over. In the real world, things are not that simple and there are various examples of non-resilient fish in simple assemblages and low-resistance fish in highly complex assemblages. The variations are due in part to the existence of keystone species (see below) and the scales at which disturbances occur. Functional groups do not all operate at the same scale, some are localised and others act over very large areas. Since disturbances are limited to specific scales, functions that operate at other scales may not be affected.

When it comes to fish chains, this has several consequences. Firstly, the number of ways a fish assemblage may recover depends on its functional groups. The impact on large fish species is of particular significance in this connection. Indeed these fish are often unique in their functional groups, which may be very important to the system, e.g., because they prey on grazers or transform the substrate. The elimination or reduction of these species can change the entire system (Jackson et al. 2001, Bellwood et al. 2003). This is referred to as the ecological inactivation of the species. It is not, however, the only lesson. Within a functional group, ecological functions are unevenly distributed among the species, with driver species making a large ecological impact and passenger species a minimal one. The addition of drivers increases the stability of the system, while passengers have little or no direct effect (Peterson et al. 1998). One of the goals in a fish chain is to increase the yield per species and to achieve this goal, passenger species are often either intentionally or unintentionally eliminated from the system. This reduces the resistance of the system and increases its vulnerability. It may become more resilient but less productive on an all-species basis.

The concepts of stability, resistance and resilience are a matter of scale. It is usually the case that the larger the scale and the larger the stability and resistance, the smaller the resilience. Historical changes are interesting from this perspective. They show that the numbers of fish species in marine ecosystems have not changed much over time, but that some disappearing species have induced drastic changes in the structure of the ecosystems (Jackson et al. 2001, Pauly et al. 2002). These changes usually take a long time, indicating that on a large scale, resistance may be important. Unfortunately, once the changes have taken place, there is very little chance of a

return to the initial state (low resilience). On the local scale, however, if some catastrophic event disrupts the state of an ecosystem, in many cases, the system bounces back. Catastrophic events may even be a necessity to some systems with high diversity such as coral reefs and tropical estuaries (Connell 1978; Blaber 1997). On the same local scale, slow but constant perturbations often cause more damage than catastrophic events because environmental conditions such as pollution, increasing fishing or major land use changes prevent the system from bouncing back.

Perturbations are necessary to maintain diversity and ecosystems maintain their diversity as long as perturbation levels and types stay within a given range. The problem facing most ecosystems nowadays is a change in the perturbation patterns and levels that have the potential of inducing long-term changes called *phase shifts*. This is due to the increasing actions of humans that affect nature. It has become increasingly difficult to separate natural from human-induced disturbances. This is illustrated by the synergistic effects of overfishing and a natural disease affecting an algae-grazing urchin in the Caribbean (Hughes 1994). This has resulted in a complete change in the benthic landscape, which is transformed from coral-dominated to algae-dominated, the results of which can be potentially disastrous for the reef fish community

Types of Diversity Change and their Consequences

Diversity may change in various ways. There can be species losses or gains usually associated with changes in evenness and variations in the functioning of fish assemblages. In a system undisturbed by humans, there may be species gains resulting from migration or speciation, both occurring on long time scales. In systems influenced by humans, alien species can be introduced, some of which may become invasive. We consider a species invasive in an area if it is known to be new there and its abundance or ecological roles are such that the system in the area is modified. Not all introduced alien species become invasive (Kolar and Lodge 2002) and not all invasive species are introduced by humans. However, the emergence of most documented invasive species follows their deliberate or accidental introduction by humans.

Migration and speciation are natural phenomena that seldom occur at time scales that affect fisheries. Their effects on ecosystems may be very important but the changes are usually gradual. Conversely, the introductions and invasions of alien species may occur over short time scales and have drastic effects on ecosystems. Most invasive alien species are initially introduced by humans either deliberately (e.g., Nile perch in Lake Victoria, common carp in the USA and Australia) or accidentally (e.g., lamprey in the Great Lakes). There are also cases of natural invasions, e.g., triggerfish (*Balistes carolinensis*) in West Africa (Caveriviere et al. 1981). Invasive species may have dramatic effects on ecosystem structures. This is well illustrated for freshwater species (e.g., tilapias in many Asian and Pacific coun-

tries, Nile perch in Lake Victoria). Introductions of alien marine organisms are increasing. They include deliberate introductions such as *Cephalopholis argus* and *Lutjanus kasmira* in Hawaii (Randall 1985) and the unintentional results of human interventions such as Lessepsian species in the Red Sea and the Southeast Mediterranean and introductions through ballast water (Wonham et al 2000). The effects of these introductions can be devastating, for example for the *L. kasmira* in Hawaii (Randall 1987), jellyfish in the Black Sea (Shiganova and Bulgakova 2000) and sea-stars in southern Australia (Ross et al 2004). However, not all introduced alien species become invasive. Most Lessepsian species have become integrated into the Southeast Mediterranean basin fish community without generating any important changes. The major problem with the introduction of alien species in the marine environment is that once they have become established or invasive, they are almost impossible to control or eradicate.

Species loss is as important to diversity and ecosystem functioning as species gain. In general, species loss is associated with extinction or extirpation, which are *true losses*. Extinction means the species no longer exists on Earth. Like speciation, extinction is a natural phenomenon, but man can cause extinction rates far beyond the average ones found in nature. Extirpation is the loss of a species over part of its native range. There is also species inactivation, which means that even though it is still present, the species has reached such a low abundance or biomass that it no longer plays a significant ecological role. Inactivation is considered a natural process because within a functional group, species tend to replace one another over time depending on the environmental and species history. However, humans may induce rates of species inactivation that are higher than normal and thus unbalance fish communities or their ecosystems.

A number of studies (Jackson 2001, Jackson et al. 2001) indicate that before human intervention, coastal ecosystems had structures very different than the present ones. Although very few marine fish have disappeared so far due to human disturbances, the number of extinctions may increase dramatically in the near future for species with a restricted range (Hawkins et al. 2000). The consequences of the disappearance of a species in a system depend on its functional group and its role in the ecosystem. The first species to disappear are usually the ones that are of high commercial or subsistence interest. These species are often large predators that play very important roles in the ecosystem. Their disappearance may irreversibly unbalance the system (Jackson et al. 2001). Species losses of marine fish are usually extirpations, and very few cases of total extinction have been documented up to now. However, extirpation can have very serious consequences, in particular if the distribution is patchy (Hanski as cited by Côté and Reynolds 2002). In the Caribbean, the disappearance of the Nassau grouper (*Epinephelus striatus*) and other large groupers of the genus *Mycteroperca* over most of their range has had numerous direct and indirect ecological impacts (Sadovy 1993; Roberts 1997). Direct impacts are reductions in the preying on small reef fish that may play an important role in structuring the reef landscape and thus, indirectly, the reef-fish assemblages. Indir-

ect impacts include shifts of fishers towards other large carnivorous fish, further depleting this functional group.

This brings us to the role of rare and endemic species in the functioning of coastal marine ecosystems. There are driver and passenger species in a functional group and the dominance of species there may change in time and space. A species' rarity is a combination of geographical range and density, since a species may be rare in one place and abundant somewhere else and its role in the system will change accordingly (Jones et al. 2002). In the marine environment, most fish species have the potential to disperse over large geographical areas. This generates lower endemism rates (Mora et al. 2003) than in terrestrial systems, even on remote islands. There is an endemism rate gradient in the tropical Pacific, with the highest rates in the central Pacific (Hawaii, Marquesas, Easter Island) and the lowest ones in the western part (Robertson 2001). Similarly, endemic species tend to be larger and more abundant in the central Pacific than the rest of the Pacific. This could be due to different causes depending on the region. In isolated areas, endemism may be the result of local speciation and in areas close to a biodiversity centre, endemic species may be relic species that used to have a wider geographical distribution. This suggests that the roles of endemic species in the functioning of fish communities probably differs from one region to another. Unfortunately very little is known about the causes and effects of rarity or the relations between endemism and abundance among marine fish (Robertson 2001, Jones et al. 2002).

In general, there is a much higher percentage of undescribed marine species in the tropics than in temperate or cold regions. In addition, the geographical range of species is far less known in tropical than in temperate or cold areas. There are more than 6100 taxa of coastal fish in the Pacific, more than 14% of which are undescribed. Most of the undescribed species are small and usually have little ecological impact. However, some large and even commercial species have yet to be described. Moreover, there is no checklist available for various regions in the Indo-Pacific. This is particularly true of a number of the Pacific island states such as the Solomon Islands, Vanuatu or Tuvalu. The situation is even worse in eastern Africa. In Eritrea, Somalia, Tanzania and Mozambique, little is known on the distribution of coastal fishes. These gaps are obstacles to understanding the status of coastal marine fish in these countries because there is a link between regional diversity and local diversity and local diversity is usually an important factor in fish density and biomass.

In addition to inadequate taxonomy, we are faced with many problems in sampling local diversity. Without going into the details of the numerous sampling technique biases, it is clear that no method can give a precise image of fish diversity (richness and evenness) in coastal waters. In general, the precision of the methods decreases with increasing diversity, since each species presents a different response to the sampling method. The biases are usually impossible to assess correctly because we have no access to what the *true* community is. In addition, most sampling techniques are adapted to one type of biotope. Since the tropics have more biotope types

and habitat heterogeneity than temperate or cold regions, multifarious sampling methods are needed to get a correct sample in the tropics. Many tropical countries have neither the means to conduct intricate sampling nor the specialists to interpret them, generally leading to less adequate knowledge of the fish diversity in many coastal tropical fisheries than in their temperate counterparts. The notable exceptions are shallow coral reef fisheries, where clear waters allow underwater visual censuses that can record a high proportion of the species present, even though the reefs support the most diverse marine fish assemblages known to man.

In addition to recording species and their relative abundance, knowledge is called for on the biology and ecology of the fish in a fish assemblage to understand how the assemblage is structured and functions. Here the knowledge gap between temperate and tropical systems is similarly sizeable. There are far more species in the tropics and far less has been invested in studying the biology and ecology of tropical fish than temperate ones. There is a lack of basic information on the growth, reproduction, mortality and movement of most tropical species. This is an obstacle for the management of tropical fisheries.

Large-Scale Variations in Fish Diversity

It is surprisingly difficult to get reliable information on the world-wide distribution of coastal marine fish. The data presented here were extracted from FishBase (Froese and Pauly 1998). However, a number of problems were encountered in defining coastal marine species and getting information on their size and diet. That is why the data are indicative of major trends but need to be refined for further analyses. The distribution of coastal marine fish diversity on a world-wide scale (fig. 3.4) indicates huge differences in diversity from one region to another (nearly ten-fold between the Northwest Pacific and the tropical West Pacific). These differences are not randomly distributed. The highest diversities are found in the tropical Pacific and Indian Ocean and the lowest in the northern parts of the Atlantic and Pacific oceans. This spatial distribution of diversity is very probably a major source of the differences in landings. In highly diverse regions, it is likely that the first 25% of the landings is composed of many species and in regions with low diversity, the same 25% probably consists of a restricted number of species. This difference could historically explain why fisheries management started by addressing the species as a management goal rather than the ecosystem. If your catch is composed of only a few species, you tend to think that by controlling these few species you can master the whole community. Because this was the situation observed in the North, where most fisheries science was initiated, it became the basis used for most fisheries management. Had fisheries scientists been confronted from the start with a very mixed catch as in the Indo-Pacific, they probably would have taken the same attitude as the local people and proposed management based on a community approach. It is not surprising that the concept of

marine protected areas (MPAs) arose precisely in regions with very high fish diversity, where management on a species basis may not have been profitable.

Despite its low level of precision, the available information (fig. 3.4) clearly indicates that, even at large regional scales, there are major differences in the characteristics of coastal fish species. This undoubtedly has important consequences as regards resource use and management. Regions where herbivorous species are diverse and large (e.g., western Pacific) are not apt to have the same response to a change in primary production as regions with a few small herbivorous species (eastern Africa) at the same latitude. There would certainly be a great deal to learn from a more detailed analysis of this regional distribution. In particular, there are probably interesting correlations between these broad-scale structures and primary production, habitat type, and the nature and level of the catch. Unfortunately, the extensive data needed for this type of analysis are not easily available because, until recently, they were not recognised as a priority in understanding fisheries.

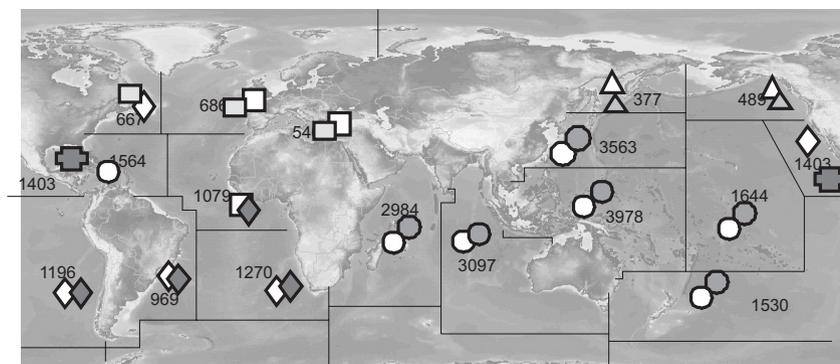


Fig. 3.4 A Distribution of coastal marine fish diversity

The numbers (A) indicate how many species are known in each region. Regions are grouped according to the size distribution or diets of their species, the symbols are as in (B). Regions were re-grouped according to a cluster analysis using Ward's method and Euclidian distance. Coastal marine fish are defined as any fish living within less than 100 metres and represent 11,280 species. The data were extracted from FishBase 2000 (Froese and Pauly 1998) and were completed by analogy, allocating the same diet to species in the same genus and of similar size.

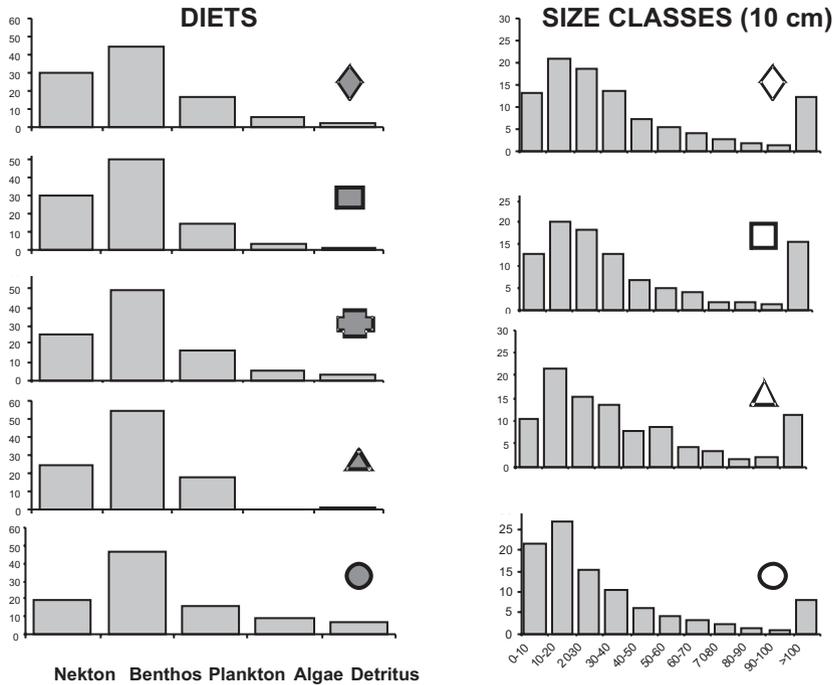


Fig. 3.4 B Distribution of coastal marine fish diversity

Virgin Systems

There are no longer any virgin marine systems. Even the most remote areas are either exploited themselves or are facing the consequences of exploitation in nearby regions. There are, however, numerous accounts of what pristine marine systems used to be like (e.g., Jackson 2001, Jackson et al. 2001). The accounts have several points in common:

- large marine organisms such as turtles, manatees, or large predatory fish used to be common and at times abundant in many systems;
- the loss of these large animals induced major ecological changes such as very heavy mortality in sea-grass beds, coral or kelp;
- some of these systems have undergone phase shifts and no longer resemble the initial systems (e.g., Caribbean reefs, Northeast Atlantic kelp forests, Chesapeake Bay) and are unlikely to have the potential to return to their initial state in the foreseeable future;
- huge natural fluctuations also occur in the absence of human intervention;
- there is often a long time lag between the start of the perturbations and their major ecological consequences;

- as fishing proceeds there is a reduction in the size of the targeted species as well as a shift from species high in the trophic organisation to species from lower trophic levels.

The most targeted large fish species in marine systems are long-lived carnivores that reproduce late in life and, in the case of sharks, bear small clutches of live young (Stevens et al. 2000). Many of these species congregate to spawn (e.g., Sadovy 1996) and are thus very vulnerable to modern fishing methods. Their features render these species very sensitive to exploitation and they are thus slow to recover (Jennings et al. 1999). They often play an important role in controlling lower trophic levels and their collapse may generate long-term changes in their marine systems (Jackson et al. 2001). However, large animals are not the only ones responsible for maintaining the major characteristics of an ecosystem. The loss of features such as coral on reefs, oyster beds in estuaries, sea-grass or algae beds can induce major ecological shifts (Jackson et al. 2001). The losses may be due to direct capturing and mechanical damage from fishing gear (Hughes 1994 on reefs; Jackson et al. 2001 on sea-grass beds and oyster beds). Disease and the loss of keystone species are also major contributors to change. Habitat-constructing organisms often need to be in high densities to maintain themselves. Once they get past a certain threshold, they quickly disappear with little chance of return. Irreversible changes or phase shifts have probably been more common than we think, since we know so little about the initial status in many areas. Even major systems such as coral reefs were barely documented before 1960. Major events occurring today on reefs such as invasions of crown of thorns starfish or coral bleaching thus have barely any historical references.

There is often a long time lag before major events start occurring. It took two centuries for cod fishing to collapse in the Northern Atlantic. The major problem is that nowadays, with the advent of increasingly sophisticated gear, collapses of this kind could accelerate. In addition, interactions at large geographical scales are occurring that we did not think possible even in recent times. The large decrease in many spawning stocks around the Caribbean islands could thus have major influences on the recruitment levels of islands far from these spawning grounds and result in a domino effect of successive collapses (Roberts 1997).

There have been a number of reviews on the effects of fishing on marine systems (e.g., Jennings and Kaiser 1998; Hall 1999; Hollingworth 2000). It is, however, essential to look at some of the major effects and how they may intervene in an ecosystem-based approach to fisheries in a North-South perspective. Fishing down the food web and catching smaller and smaller fish are common to most marine systems. However, the scale of these changes is often difficult to assess without historical documentation. There are very few places where long time series can be observed and we all too often look at the existing system as the reference point and analyse changes from a short-term perspective.

Direct Impacts

a. On substrate and benthos

Most fishing gear has some impact on the environment. Trawl nets and dredges are probably the most widely studied gear in this respect (e.g., Moran and Stephenson 2000). The effects of trawl nets on soft or rocky bottoms can be devastating, especially if the gear is fitted with tickler chains or rock-hoppers. Studies on the northwest shelf of Australia (Sainsbury 1987; Hutchings 1990) indicate that most of the epifauna that initially harboured prime target species were destroyed by a few years of trawling, resulting in a shift of fish species and a large decrease in the epifaunal diversity. In addition to killing sessile epifauna, trawls and dredges kill invertebrates (echinoderms, molluscs, worms, crustaceans and so forth) that are food for fish and they have a mechanical effect on the sediment (bioturbation). In addition, the gear has a mechanical effect on the sediment itself. Several studies show a compacting effect (e.g., Schwinghamer et al. 1996). This results in a loss of diversity and major changes in the structure of the benthic fauna (Hall 1994) as well as a homogenisation of the fauna, flora and their substrate (Brand et al. 1991). The gear also increases water turbidity and re-suspends sediment, which can modify the primary production with multifarious potential effects on the epifauna and their predators (Caddy 2000). Sea-grass beds can be affected by both the mechanical effects of the gear and the increased turbidity (Fonseca et al. 1984).

There may be recovery from the gear and the rate of recovery is a function of the natural rate of disturbance. In areas where the sediment is often naturally disturbed (e.g., estuaries), the effects of trawling may not be drastic. Conversely, trawling over deep-water sea-mounts may have very long-term effects. Slow-growing and late-reproducing organisms are more affected than short-living ones which, as several studies indicate, can recover in less than a year. However, especially in deep waters, recovery can take a very long time for long-living organisms, in some cases decades or more. The lengthy recovery time for large epifaunal organisms may be a major problem, since they are often at the base of microcosms that drastically increase the diversity in otherwise poorly diversified habitats. Despite the effects of trawling on the abundance, species composition and size structure of benthic invertebrate communities, their trophic structure seems rather stable (Jennings et al. 2001). In the tropics, several specific fishing techniques are known to damage habitats and their fauna or flora, e.g., drive nets, poison and explosives (Maragos et al. 1996), brush parks or juvenile shrimp nets (Blaber et al. 2000).

b. On fish

The direct effects of fishing on fish or other resources start at the species level with a decrease in abundance and biomass, a shift towards smaller sizes, increased mortality and growth and reproduction alterations. These variations have been mainly studied at the population rather than the community level. Fishing and pollution are the major causes of fish diversity change (Jennings and Kaiser 1998). As is noted above, there are very few cases of documented extinction among marine fish (Powles et al. 2000). There are, however, many examples of extirpation and ecological inactivation. Fishing can also significantly reduce species density in tropical fisheries (e.g., Jennings and Polunin 1997; Watson and Ormond 1994; Roberts and Polunin 1992, 1993). Decreases in diversity in temperate waters are not easily demonstrated (e.g., Greenstreet and Hall 1996). The differences may stem from the type of biotopes analysed. On tropical reefs, many species are site-attached and long-lived and have restricted adaptation to change. They may thus be quite sensitive to fishing or its indirect effects. The temperate fisheries tested for changes in diversity are all trawling grounds where most species are migratory to some extent, their life span is usually short in comparison with tropical reef fish and they adapt quickly to new environmental conditions. It is likely that a survey of tropical trawling grounds would produce similar findings, i.e., relatively minor changes of diversity over time despite intense fishing as suggested by data presented by Bianchi et al. (2000). This suggests that perhaps we should not base our judgement of the effects of fishing on diversity on a stable versus variable environment gradient rather than a tropical versus temperate to cold gradient.

In stable environments, species are expected to be rather long-lived, have limited flexibility in their life-history traits and form part of diversified functional groups with high functional similarity (Martinez 1996). In the tropics, reefs are the typical stable environment. At all latitudes, sea-mounts are also candidates, as are rocky shores or deep-water coral banks in temperate and cold climates. Conversely, variable environments such as the open ocean (pelagic fish communities), estuaries and to a lesser extent soft-bottom continental shelves can be found at any latitude, with of course an increasing diversity in warmer climates. In these environments, most species have shorter life cycles and more adaptable life histories and on average functional groups have fewer species. Stable environments tend to resist but to have little resilience. In extremely stable environments such as sea-mounts or deep water fisheries, resistance may be weak since recruitment, growth and production are very low because of very low input in these systems. Diversity loss in terms of species density can be rapid and the system can be very slow to recover (Koslow et al. 2000). The opposite holds true for the least stable environments (e.g., pelagic systems in upwelling regions), where resilience is usually very important.

The impact of fishing on density or biomass is drastic at the species level but not necessarily at the community level. The total fish production of the

North Sea was fairly stable over time (before 1980), even if most species exhibited wide temporal fluctuations in their landings (Ursin 1982). In other words, global fish production remains approximately the same from year to year but the species involved may change. It is likely that fishing is not the only cause of fluctuation in many species, and in many cases recruitment variations are probably just as important. This hypothesis of rather stable global production needs to be better substantiated and many fisheries seem to point the other way, i.e., to a decrease of global production beyond a certain level of fishing effort. This reasoning has to go along with the notion of fishing down the food web. In a mature system, production is minimal and the highest trophic levels make an important contribution to the total biomass. Once exploitation starts, the largest and often least productive individuals are taken out first. As exploitation increases, the community consists of younger individuals and gradually of species at lower trophic levels. As a consequence, global production should increase. However, the system reaches a point where individuals are caught at their minimum capturable size and only the lowest trophic levels remain in sizeable quantities. With further exploitation, either the system collapses or the production remains rather stable. In heavily exploited systems, another likely consequence is the possible dominance of the catch by fewer and fewer species as suggested by the data presented by Bianchi et al. (2000).

The effects of fishing on the structure of fish communities also need to be considered. Their structure is usually stratified into several components, the two most common ones being trophic and size structures. However, this view is simplistic, just as trophic chains are a simplistic view of trophic webs. Structure comprises many other aspects such as behaviour or reproduction. The various components interact. A large piscivore with a wide home range bears very little resemblance to a small sedentary piscivore, even though they share the same trophic level. The two species do not have the same impact on the system, nor are they affected by the same factors. Until recently though, they were often pooled together in the analysis of community structures. Several authors address this problem (Kulbicki 1992; Thiebaut and Dickie 1993; Duplisea and Kerr 1995; Garrison and Link 2000 (see www.ird.sn/activites/sih/symposium/Programme.htm)).

As most fishing gear is size-selective, fishing induces a decrease in size in many species at multi-species fisheries. This decrease is more acute in large long-lived species than small short-cycle ones (e.g., Zwanenburg 2000). Do the changes result in an increase in total production and thus in potential yield? Does the system stabilise after sufficiently long and intense exploitation? Comparison with terrestrial ecosystems gives the impression that production should increase as trophic level drops and size structures evolve towards smaller specimens. However, marine systems have several specificities, including the link between recruitment and environmental conditions. As Jennings and Kaiser (1998) conclude in their review, 'Most of the marked effects of fishing on diversity and community structure occur at relatively low levels of fishing intensity. However, once systems enter

a fished state, diversity and overall production may often remain relatively stable despite further changes in fishing intensity'.

Fishing may also affect the life-history strategies of species. Fishing eliminates the largest specimens in a population, which can have drastic effects on genetic diversity, growth, reproduction, behaviour and mortality. There is still very little information on the genetic changes induced by fishing at the population level (Law 2000), the best studied cases being the anadromous salmonids (Ryman et al. 1995). Fishing tends to reduce heterozygosity, but there are few documented cases of the impact of fishing on genotypic diversity, one exception being the study by Smith et al. (1991) on the orange roughy (*Hoplostethus atlanticus*) found off New Zealand. These authors demonstrate that even after a relatively short exposure to fishing, the heterozygosity of fished populations of this species is strongly reduced. Responses of fish to fishing can be genotypic as well as phenotypic and it is usually difficult to separate the two sides of life-history traits (Stokes et al. 1993). In heavily fished areas, growth can be phenotypically enhanced for small fish since the larger individuals are no longer at the top of the pecking order and food availability is higher. Similarly, fishing may genetically select in favour of faster growing fish since they reach their full size and maturity earlier and are more likely to reproduce than slower-growing, later-reproducing individuals. Rijnsdorp (1993) demonstrates that changes of this kind observed in plaice in the North Sea are more likely to result from genotypic than phenotypic variation linked to fishing. More recently, Conover and Munch (2002) demonstrate experimentally that fishing significantly affects growth on a genetic basis. The ramifications of their results are debated (Malakoff 2002), but they suggest that management measures such as MPAs directed separately at juvenile and adult phases could be effective.

In many tropical species, there is a change of sex with size, some species being first female and then male (e.g., wrasses, parrot fish, groupers) and others vice versa (e.g., *Centropomidae*, *Synodontidae*). Traits like this are also observed in temperate or cold-water species but to a much lesser extent. Fishing for larger individuals can thus modify the sex ratio in populations of these species and sometimes to a dangerous level (Thompson and Munro 1983; Sadovy 1996).

Beyond the effects of fishing on the genetic characteristics of the resource, there is an increasing awareness of the importance of genetics in fisheries and related fields. Bowen (1999) discusses the policy implications of conservation at gene, species and ecosystem levels. The American Fisheries Society's publication on evolution and the aquatic ecosystem (Nielsen and Powers 1995) is a milestone in its provision of information on the application of genetics in fisheries management. Since then, applied genetics in fisheries management has expanded very rapidly, accompanied by expanding recognition of the need to fill gaps in fish genetic resource policy and take appropriate action (e.g., Harvey et al. 1998; Pullin et al. 1999). In addition, changes in genetic diversity are becoming easier and cheaper to monitor and the genetics of exploited fish populations and living compo-

nents of their supportive ecosystems are likely to be the focus of far more attention in fisheries management. This should make it possible to carry out the proposed efforts towards more effective management of fish genetic resources (Pullin 2000).

Indirect Impacts

Fishing can have numerous indirect effects on marine habitat structure, fauna and flora (Blaber et al. 2000). In temperate regions, the most well-known changes are the ones affecting sea urchin densities (Tegner and Dayton 2000). In Alaska, changes of this kind have been correlated with killer whales preying on sea otters, and on the Northwest Atlantic coast they were first attributed to the reduction of predation by lobsters (Mann 1982). Subsequent studies demonstrate that lobsters are probably not sufficient enough to control the urchins and other causes such as variations in urchin recruitment can explain the changes (Hart and Scheibling 1988). In tropical regions, the most illustrative examples are linked to the decrease in predation on urchins generated by fishing for urchin predators (McClanahan 1994). In Kenya, fish that feed on urchins are removed by fishing to the point where urchins proliferate and decrease algae by grazing. However, the same causes do not necessarily have the same effects. Recent studies by SPC and IRD (unpublished data) in the South Pacific fail to show any correlation between fishing intensity and urchin abundance at various spatial scales, even though fishing is intense in several places and the species composition exhibits many similarities with the Indian Ocean ichthyofauna found in Kenya. There are many other unexpected and indirect impacts of fishing, e.g., the increase of litter in mangroves next to crab fisheries and changes in the trophic structure of West African estuaries next to brush parks (Blaber et al. 2000).

In temperate regions, many studies investigate the relations between fishing and the top predators represented by sea birds (Tasker et al. 2000) and marine mammals (Jennings and Kaiser 1998). The reproductive success of some sea birds is heavily dependent on the abundance of small coastal pelagic fishes. The fish are dependent on combinations of environmental conditions and fishing mortality. Where these fish populations have markedly declined, so have the sea bird populations, e.g., in Peru, South Africa and the northern Atlantic. The rejection of trash fish and other by-catch has however boosted populations of scavenging sea birds, mainly sea gulls. Declines in several seal populations have coincided with increases in fishing efforts or changes in the target species by fishers, with corresponding declines in seal prey, e.g., in Alaska, Peru and northern Europe. In tropical countries, this concern for sea birds or mammals may at first seem trivial. Neither sea birds nor marine mammals are abundant in most of the tropics and fisheries managers usually do not devote a great deal of attention to them. For several reasons though, this is likely to change. Firstly, there is an ever-growing increase in the import of marine products from

tropical countries by temperate developed ones. Public opinion in developed countries is increasingly sensitive to animal rights, particularly those of charismatic species including sea turtles, marine mammals and some sea birds. Products from countries where no attention is devoted to these animals might be boycotted, as was the case with Mexican tuna when the Mexican tuna fishing industry failed to comply with US regulations on porpoises. Secondly, these top predators may play an unforeseen ecological role in the functioning of the systems. Thirdly, some marine mammals, in particular whales, are extremely valuable for tourism. Taking the ecological needs of these animals into account could enhance tourism in places where it is an important source of external revenue (e.g., Tonga, Fiji, and New Caledonia in the Pacific).

Huge quantities of fishing bycatch are discarded every year, an estimated 27% of the world's total catch (see Britton and Morton 1994 for a review). In addition, many kinds of fishing gear, for example trawling, blast and poison fishing, kill fish and benthic organisms that are not brought to the surface (side-kills). Bycatch and side-kills are eaten by sea birds, marine mammals, and bottom-dwelling organisms. Numerous studies show that bottom-feeding fish and crustaceans feed on these food sources, with at times huge increases in abundance. Changes like this in densities may unbalance the systems they occur in. Very little research has been conducted on bycatch and side-kills in tropical countries. For several reasons, they could play a different role than in temperate or cold systems. Firstly, bycatch is usually minimal in tropical countries, where people find a use for most marine products. Secondly, sea bird and marine mammal populations are usually less abundant there (except in coastal upwelling systems such as Peru) than in temperate countries, so that a higher percentage of the discarded catch should reach the bottom. Along with side-kills, this material becomes prey for a much higher diversity of benthic scavenger organisms on the bottom than in temperate or cold regions. Its availability per scavenger species should thus be lower in the tropics. Moreover, decomposition is faster in the tropics and tropical carrion feeders, sharks excepted, tend to be smaller than in cold and temperate waters.

Interaction of fishing with other disturbances

Large-Scale Disturbances

Very few studies have been conducted on the possible consequences of global change and capture fishing. Zwanenburg (2000) considers the potential effect of bottom-water temperature rises off the Scotian shelf (NW Atlantic), which might be linked to global changes. There are also numerous reports of warm-water species recently observed out of their usual range, such as the trigger fish *Balistes carolinensis* in the northeastern Atlantic. Conversely, some cold-water species have begun to disappear from the

warmest parts of their distribution ranges. Temperature shifts associated with global warming are believed to be potentially important in changing some major current patterns such as the Gulf Stream and in raising the sea level. Changes in current patterns could have huge and unforeseen effects on the recruitment of most types of fish with either pelagic eggs or larvae. Changes of this kind are well documented along the coast of Peru where El Niño, a natural phenomenon, causes drastic changes in the recruitment patterns of small pelagic fish with multifarious effects on their predators and on fisheries. The rise of the sea level could have dramatic effects on many estuarine systems and coral reefs. However, it should be noted that if such a rise occurs, it would probably be rather slow. Some corals and mangroves would have time to adapt to this type of change. The apparent increase in catastrophic events such as tropical storms (Done 1999), crown of thorns starfish infestations and coral bleaching is a matter of greater concern in the tropics. The variation in the frequency of these events is thought to be linked to global change. Tropical storms are known to destroy coral over large expanses (Scoffin 1993; Dollar and Tribble 1993) and to be major factors in coastal systems associated with estuaries (Blabler 1997, Chapter II). Coral bleaching and crown of thorns starfish infestations can similarly lead to the mass destruction of coral (Hoegh-Guldberg 1999; Nyström et al. 2000). The consequences are still not well understood, but it is likely that reef- or estuarine-associated fish diversity will decrease (McManus et al. 2000), with a probable increase of herbivores in reef communities. There could also be phase shifts on small isolated islands where recolonisation by coral may be slow.

Local Disturbances

Local disturbances are likely to change coastal marine resource systems. The changes can be drastic and not necessarily limited to the immediate disturbance area. As indicated by Caddy (2000), the effects of high nutrient inputs from rivers in the Mediterranean region extend far beyond the river mouths. Similar effects are observed on a large-scale in the Baltic Sea, where the overall fish biomass is thought to have increased four-fold in the twentieth century (Thurrow 1997) due to terrestrial nutrient inputs. This demonstrates that disturbances can have even more profound effects than intense fishing. The opposite effects are observed in the Black Sea, where the deep anoxic layer is gradually becoming shallower due to the inputs of rivers such as the Danube, the Don and Dniepr and is jeopardising the entire ecosystem (Caddy 2000). Another infamous case is the slow death of the entire Aral Sea from the pumping of the Amou-Daria drainage system water into it as a result of the cotton culture. In the tropics, similar effects can be expected from the extensive logging in many areas (e.g., Central Africa, Amazonia, Indonesia, Papua New Guinea, Solomon Islands) or open pit mining (e.g., Indonesia, New Caledonia, Fiji).

A recent addition to the localised coastal disturbances is the advent of intensive aquaculture in coastal zones. Shrimp farms have destroyed huge areas of mangroves in Southeast Asia and Ecuador and the high quantities of nutrients in their wastes have led to sizeable amounts of pollution (e.g., Suvapepun 1997). The destruction of wetlands and mangroves is a major concern in many countries around the world. These areas are often of paramount importance in the cycling of many coastal resources (Blaber 1997; Blaber et al. 2000). However, the importance of mangrove areas for reef or soft-bottom fish species is variable from one region to the next (Thollot 1992). In particular, the Caribbean and southwestern Pacific mangrove systems play very different ecological roles as regards reef fishes.

Urban development and coastal zoning may have important impacts on coastal resources by modifying a number of cycles and recruitment, as well as increasing pollution and nutrient inputs. In most cases, this results in a decrease in the habitat quality and a subsequent decrease in the diversity of the resource and non-target species, with subsequent decreased resistance and often with increases of *r*-selected species. Tourism also exerts a substantial influence on coastal resources (e.g., Maragos et al. 1996). It increases the demand for marine products, especially those that fetch high prices (e.g., crustaceans, large carnivorous fish). It also emphasises the need for pristine areas (e.g., marine parks) and the need to protect charismatic species (e.g., turtles, seals, whales and porpoises). Tourists can cause direct damage to marine habitats by trampling on corals fringing reefs, turning over rocks in search of shells and so forth.

This chapter highlights the roles of diversity and environmental factors in marine fisheries. As regards diversity, we have changed our philosophy of sampling for the purposes of understanding and governing fisheries. Most of the historical data have only been collected for target species. The taxonomy of non-target species is problematic, even in areas such as the North Sea (Bianchi et al. 2000, Vecchione et al. 2000). Diversity assessment is very sensitive to the methods and it is important that standardised methods be used. As Bianchi et al. (2000) note in their conclusions, 'Efforts should be put into standardising data collection and developing appropriate sampling design to satisfy requirements of comparative studies'. Standardisation is often possible on small scales if only a few collectors are involved. However, it is a much greater problem on a regional scale, since there is no international standard. Efforts are currently underway in the South Pacific to standardise reef fish sampling procedures (Kulbicki et al. 2004), but it is difficult. It becomes even more difficult if the species to be surveyed are not accessible by the same method or there are huge interspecific biases within the method. In an ecological approach, we might for instance wish to consider all the fish in a lagoon with mangroves, soft bottoms, and reefs. At present, there is no method that can sample all three biotopes accurately because each biotope requires its own specific method.

Diversity can also be considered at higher levels than taxa. Functional groups can play a major role in improving our understanding of ecological processes, since species replacement in a functional group is not uncom-

mon (Jennings and Kaiser 1998). This approach requires that functional groups be clearly defined and that we have enough information to classify taxa properly. At present the concept of the functional group is still fuzzy and there is no unequivocal definition of what it is or is not. This is probably not a problem in simple systems, but in complex ones as in most tropical coastal marine environments, the issue can quickly become difficult. This is even more so due to the lack of information on the life-history traits of species. There is less and less incentive for scientists to publish work on basic traits such as diet, reproduction (behaviour, sex ratio, size at first maturity and so forth). This type of work is increasingly regarded as descriptive and is thus not well accepted by many scientific journals. This kind of information is however essential to defining functional groups and to trophic analyses, which are increasingly considered an essential step in understanding ecological processes for management purposes (Jennings and Kaiser 1998; Pauly, Christensen and Walters 2000).

Environmental factors should also be better integrated into fisheries governance. It is not so much that more environmental measurements are needed as that they need to be more effectively used to better understand ecological processes. The environment has a paramount influence on the ecological processes governing coastal marine resources. Of course we need to improve the ways we acquire and process environmental data, but that should not be the most important point. We need to relate more accurately resource and environmental data sets to the appropriate time and spatial scales.

One important point that is clear from a review of the current literature on the ecological aspects of fisheries is that there are so few possible control areas for comparison. In most regions, fishing has been going on for so long and so intensively that there are no control areas against which to compare their current status. Jennings and Kaiser (1998) suggest creating reserves to give us an idea of the status reached by resources in undisturbed or less-disturbed areas. The study of islands, especially in the Pacific, may in part solve this problem for reef resources since there is an entire gradient in the disturbance level of islands to allow a comparative approach.

From a North-South perspective, it is important to note that diversity and complexity tend to be greater in southern ecosystems. Sampling the higher diversity and complexity of ecosystems in the South is problematic. In addition to the inadequate taxonomy and information on biological traits in the South compared to the North, coastal marine resources in the South exhibit greater spatial and temporal heterogeneity. One consequence of this heterogeneity is a need for better sampling strategies and higher sampling efforts to achieve approximately the same power of analysis as in the North. There is also far less historical and reliable data for the South, which is a problem when it comes to understanding the role that is played by present perturbations and natural phenomena. Acquiring data on a regular basis (observatories) is often a problem for countries in the South due to political and economical instability, the lack of permanent structures, and rapid turnover of qualified staff. These difficulties have raised the question of the mini-

mum data needed for management decisions (Johannes 1978; Walters 1998).

Collecting data is at one end of the decision chain and management decisions at the other. This indicates a need for a framework. Until recently the approach was based on population dynamics, with target species as the focus of management decisions. The focus is now shifting from the species to the ecosystem level (Botsford et al. 1997; Beamish and Mahnken 1999; Maltby 1999; Prins 1999). At the moment, however, we do not have a unifying theory to enable us to understand the functioning of these coastal marine ecosystems. We are still at more or less a descriptive stage, trying to link ecological processes with a whole range of factors. Our lack of historical data, the problems related to controls in an experimental or comparative approach and the huge complexity of marine systems, especially in the tropics, do not make the task any easier. In addition, even if we do understand the functioning of these systems, management will not necessarily follow. The general state of an ecosystem, the level of its fish populations, fishing efforts (number of fishers and amount of gear in use, number of boats) and economic factors do not operate at the same time scales. Fishing fleets tend to increase when fishing conditions are good, but they do not diminish at a similar rate when the yields drop or market conditions are unfavourable.

Even if we think we understand ecological processes (e.g., Cury et al. 2003 for a recent classification) or if keystone species are identified, we may not be able to manage an ecosystem sufficiently to achieve the desired results. There are also major North-South differences, since increasing fishing activity to maximise the yield is less and less of a priority in the North now that other uses mainly related to conservation and recreation are becoming more important. In the South, subsistence fishing is still paramount in many places and conservation issues may seem trivial unless people can be convinced they are essential. This brings up the problem of education and understanding measures. A measure is more apt to be accepted if its consequences are understood. Basic education in ecology can prevent some tragic errors. Local people usually have ample knowledge of their own environment, but may ignore key information or have erroneous ideas about major ecological processes in their fishing area. So it is essential to include local people in the protection of their resources and take their traditional knowledge into account. This might require a specific kind of education to enable the local people to better understand the consequences of modern fishing in a monetary economy.

Promoting diversity from a governance point of view is a difficult task that involves several decision levels. It ranges from direct promotion, enhancing the juvenile survival of fish, reducing the fishing pressure on spawning grounds or limiting the use of non-selective gear, to indirect action ruling out destructive fishing methods, encouraging habitat restoration (wetlands, mangroves, near-shore reefs) or increasing habitat diversity (e.g., artificial reefs). If regulations are to be accepted, it is also necessary to promote the ecological education of various resource users. This is usually

a long and costly process. Teaching governance with an ecological perspective means helping people understand that ecosystems are complex and preserving only part of them is often less efficient than protecting an array of biotopes. The interaction between ecosystem parts or between whole ecosystems may be geographically huge, and in order to integrate the various spatial and temporal scales of interaction, governance will have to adapt from the local level all the way to the international level.

MPAs may illustrate the need to expand from the local to the regional level. This concept was initially a protective measure addressing local problems. The question was soon posed as to the size, shape and ecological complexity (i.e., number of biotopes) MPAs should have. This responds to several governance problems: 1. What do we need to protect? 2. How much can we protect without affecting other users? 3. Who is involved in the decision-making process? The size of the proposed MPAs has grown with the awareness of multiple-interactions in ecosystems and there are now even proposals for MPAs crossing international boundaries.

It is essential for improved governance to involve various resource users in the decision chain. For users to take part and accept decisions, it is essential that they understand the consequences. This means that educating people and demonstrating the effects of governance should be part of governance. In particular, education should include basic ecological information. For example, fishers usually want to increase their catch value. They can do so in several ways, by catching more fish, catching fewer but larger fish or catching more valuable species. These strategies have different governance implications. A larger number of fish means an increase in production and usually in fishing effort. This increase is easier if the number of available species is low, but low diversity systems are usually more vulnerable to perturbation. This is the problem facing most coastal pelagic fisheries, where misunderstanding the ecological processes involved in increased production has led to numerous fisheries collapses. Catching fewer but larger fish means fish are allowed to grow and juveniles and the reproduction and habitats for juveniles and reproductive stocks are protected. In many Pacific island countries, the expansion of the live reef fish trade and aquarium fish trade is an example of catching more valuable species. This implies difficult choices. Destructive fishing methods bring fast cash over a short time period, though better fishing practices yield fewer but better fish and allow a longer exploitation of the resources, even if the immediate cash flow is lower.

Making decisions and evaluating the consequences of governance decisions require measurements that can be easily understood by all the actors. Developing indicators is in part a solution to this problem and a great deal of research is currently being conducted in this field. The present trend is to propose an array of indicators ranging from ecological to socio-economic aspects of fisheries and bridge the various indicators into warning systems with a capacity to focus on the potential interaction between the various aspects.

The parties responsible for fisheries governance face some environmental factors they may directly influence and others they should take into account but have little or no influence on. Handling the two kinds of environmental factors can be viewed as *active* and *passive* governance. The climate and region largely determine the diversity, habitat types and the characteristics of the resources. Taking regional or climatic differences into account is very important but is a passive decision. Conversely, one may act directly on many factors such as fishing levels and gear, pollution, coastal and land management and so forth, but this active governance requires recognition of the consequences of intervention that can result in drastic ecosystem changes. In highly diverse systems, mainly tropical ones, resistance to change is generally high but resilience is low, whereas the opposite is often true in less diverse systems. This means changes in governance take longer to be effective in diverse systems but their effects last longer as well. In highly degraded but still diverse systems, this could mean some governance issues are hard to accept because users may not detect rapid changes. Similarly the scales involved in diverse and non-diverse systems may be different and the spatial patchiness of many diverse systems can render them less sensitive to changes than less diverse but more homogeneous systems.

Two approaches to ecosystem governance are currently recognised, a holistic one addressing the system as a whole, and a reductionist one considering each species separately and just viewing the ecosystem as a support for the species of interest. The second view has prevailed to date, but there is increasing awareness that the first approach can be fruitful and that a combination of the two with a balance of active and passive governance might be preferable. Lastly, one should consider time and space interaction with top-down governance decisions that extend from long-term and large-scale issues to shorter-term and smaller-scale issues and bottom-up governance decisions that extend from local to regional issues.

FISH FOR LIFE

Interactive Governance for Fisheries

*Edited by Jan Kooiman, Maarten Bavinck,
Svein Jentoft and Roger Pullin*

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