The fishery effects of marine reserves and fishery closures

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Front cover: Line fisherman fishing within the Soufrière Marine Management Area, St. Lucia, West Indies. A network of marine reserves increased local catches between 46% and 90% over five years of protection from fishing.

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Part 1: Review

1. Summary

Marine reserves, areas permanently closed to all fishing, are frequently proposed as a tool for managing fisheries. Fishery benefits claimed for reserves include increases in spawning stock size, animal body size, and reproductive output of exploited species. Reserves are predicted to augment catches through export of offspring to fishing grounds, and spillover of juveniles and adults from reserves to fisheries. Protection of stocks and development of extended age structures of populations in reserves are argued to offer insurance against environmental variability and management failure. Models also suggest reserves will reduce year-to-year variability in catches, and offer greater simplicity of management and enforcement. Reserves are predicted to lead to habitat recovery from fishing disturbance which can also enhance benefits to fisheries.

Extensive field research confirms many of these predictions. Reserves worldwide have led to increases in abundance, body size, biomass and reproductive output of exploited species. Such measures often increase many times over, sometimes by an order of magnitude or more. Population build up is usually rapid with effects detectable within 2-3 years of protection. Increases are often sustained over extended periods, particularly for longer-lived species and for measures of habitat recovery. Reserves have benefitted species from a wide taxonomic spectrum that covers most economically important taxa, including many species of fish, crustaceans, mollusks and echinoderms.

Encouraged by these results, many countries and states have embarked upon initiatives to establish networks of marine reserves. However, reserves remain highly controversial among fishers and fishing industry bodies who argue that fishery benefits remain unproven. In the last three years there has been rapid growth in the number of cases where fisheries have been shown to benefit from reserves. In this report, we critically analyze this body of evidence, drawing upon studies of reserves and fishery closures. Fishery managers have long used fishery closures, areas temporarily closed to fishing for one or more species or to specific fishing gears. They are employed to help rebuild depleted stocks, reduce gear conflicts, protect vulnerable life stages of exploited species or protect sensitive habitats from damaging gears. Such areas can tell us much about the potential effects of marine reserves.

Fishery benefits from reserves and fishery closures typically develop quickly, in most cases within five years of their creation. Perhaps the most persuasive evidence of fishery effects of reserves comes from changing fishing patterns. In most places where well-respected reserves or fishery closures exist, fishers tend to move their fishing activities closer to their boundaries. Fishing-the-line, as it is called, allows fishers to benefit from spillover of animals from reserves to fishing grounds. There are now well-documented cases of spillover from more than a dozen countries and including a wide range of species. It is more technically demanding to prove fishery enhancement through export of offspring on ocean currents. Existing reserves are generally small, making it hard to detect increased recruitment to fisheries at a regional scale. However, there are now several cases in which export of eggs and larvae have been confirmed, including dramatic enhancement of scallop fisheries in Georges Bank and clam fisheries in Fiji. Small reserves have worked well and repeatedly produce local benefits. However, regional fisheries enhancement will require more extensive networks of reserves. Some of the most convincing success stories come from places in which between 10 and 35% of fishing grounds have been protected. In several cases there is evidence that yields with reserves have risen to higher levels than prior to protection, despite a reduction in the area of fishing grounds. In other cases, smaller reserves have stabilized catches from intensively exploited fisheries or slowed existing rates of decline.

We describe experiences that prove that success of marine reserves is not contingent on habitat type, geographical location, the kind of fishery involved, or the technological sophistication of management. Reserve benefits are not restricted to habitats like coral reefs, or to artisanal fisheries, as some critics claim. Fishery benefits have been demonstrated from reserves established in tropical, warm- and cold-temperate waters, and in many habitats, including coral reefs, rocky reefs, kelp forests, seagrass beds, mangroves, estuaries, soft sediments, continental shelves and deep sea. Reserves and fishery closures have worked well for a wide range of fisheries, spanning recreational fisheries, artisanal fisheries like those of coral reefs, through small-scale nearshore fisheries for species like lobsters, up to industrial-scale fisheries for animals like flatfish and scallops. They have worked across a similarly broad spectrum of management sophistication, from self-policing by committed fishers, through warden patrols to satellite monitoring of distant fishing activities.

We now have strong evidence that with the support of local communities, marine reserves offer a highly effective management tool. However, reserves will only rarely be adequate as a stand-alone management approach, although we describe cases where they have worked in the absence of other measures. They will be most effective when implemented as part of a package of limits on fishing effort and protect exploited species and their habitats.

2. Introduction

Marine reserves are areas of the sea that are permanently closed to all fishing. They have been attracting the attention of conservationists and fishery managers since the mid-1980s when the first studies of their effects

began being published. Those studies indicated that protection from fishing could lead to rapid and dramatic changes in populations and habitats in reserves. As interest in reserves has developed, the number of reviews giving evidence for their effects has escalated with new ones being published every few months (PDT 1990, Roberts and Polunin 1991, 1993, Dugan and Davis 1993, Rowley 1994, Bohnsack 1996, 1998, Attwood 1997, Guenette et al. 1998, Murray et al. 1999, Jennings 2000, Crowder et al. 2000, Mosquera et al. 2000, Palumbi 2000, Halpern and Warner 2002, Russ 2002, Polunin in press, Roberts in press). There have been similar numbers of books, reports and national assessments devoted to them (e.g. Ballantine 1991, MRTG 1997, Yoklavich 1998, Fujita et al. 1998, Roberts and Hawkins 2000, NRC 2001, Birkeland and Friedlander 2001, to name a few).

Previous reviews present compelling evidence that marine reserves are effective conservation tools. Throughout the world they have had remarkably consistent effects. Protection from fishing leads to rapid build-up of abundance and biomass of populations of exploited species, increases average body size, and extends population age structure. It also increases species diversity, fosters habitat recovery from fishing disturbances, and allows the development of assemblages of species and habitats that are different from those in exploited areas (for examples see Roberts and Hawkins 2000, NRC 2001, Halpern and Warner 2002, Halpern in press).

The predicted benefits of reserves for adjacent fisheries depend on reserves protecting spawning stocks and vulnerable life stages. In short, protection from fishing allows exploited species to live longer, grow larger and become more numerous, all of which increase reproductive output. Since most exploited marine species have pelagic egg or larval stages (Palumbi 2000, Shanks et al. in press), the offspring of protected animals can be dispersed widely from reserves to re-supply fishing grounds. Furthermore, as stocks build up in reserves, there is predicted to be movement of juveniles and adults from protected areas to fishing grounds, termed 'spillover' (Bohnsack 1998). At the management level, reserves are predicted to be simpler and easier to implement in complex multispecies fisheries than single species approaches (PDT 1990). Models predict that populations in reserves will provide insurance against management failure in surrounding fishing grounds (Lauck et al. 1998), and that reserves will reduce year-to-year variability in catches, making management easier and fishers' incomes more predictable (Sladek Nowlis and Roberts 1999). Given the declining state of many of the world's fisheries, and the seeming inability of present management approaches to prevent them (Roberts 1997, 2000a, NRC 1999), the possible fishery management benefits of reserves have been attracting great interest.

Evidence for the success of marine reserves has led to growing efforts to establish more of them worldwide. The Philippines has more than 40 reserves that are well protected and hundreds more have been established there (Pajaro et al. 1999). The State of Victoria in Australia declared 13 marine reserves encompassing 5.3% of state waters in 2002 (http://www.nre.vic.gov.au/). The Galapagos Islands of Ecuador were declared a marine park in 1998 that encompasses 133,000km^2 of ocean. Eighteen percent of the coastline is included in marine reserve zones closed to fishing (http://www.darwinfoundation.org/marine/zoning.html). In the United States, two expert panels of the National Research Council have recommended marine reserves be used widely to protect species, habitats and help sustain fisheries (NRC 1999, 2001). President Clinton issued an Executive Order in May 2000 charging his agencies to begin developing a national system of marine protected areas of which a significant part would be protected from all fishing (NRC 2001). The state of California is currently embarked upon a process to create a network of marine reserves in state waters.

As efforts to set up reserves gather pace, claims for their benefits to fisheries are coming under intensive scrutiny. Proposals to establish marine reserves are often met with hostility by fishing communities. Indeed, opposition from fishers is one of the principal barriers to reserve establishment. Fishers are often wary of reserves because they see them as taking away from their livelihoods rather than contributing to them. These negative perceptions are reinforced by long experience with a growing body of regulations that have failed to halt fishery declines and fishers are not convinced that reserves will be any different. Sceptical fishers and their industry bodies are demanding proof that reserves can deliver fishery benefits before accepting them, while others are showing outright hostility towards reserves. For example, in the United States, recreational fishing industry groups are opposing reserves and supporting a bill being considered by Congress, The Freedom to Fish Act, which would greatly limit the ability of regulatory agencies to protect areas from fishing (http://www.freedomtofish.org/f2f/f2f_act/#house).

Recently, there has been a burst of new evidence concerning the fishery effects of reserves, warranting this review. In it we synthesise this evidence and focus our analyses around a series of questions about reserve performance frequently asked by fishers and fishery managers. We complement our review with a series of case studies documenting reserve effects from all over the world. Readers seeking a greater level of detail should refer to them. In preparing these studies we have benefited from the generosity of many researchers who have shared their latest findings with us. We demonstrate that the case that reserves benefit fisheries has become much stronger over the last two to three years and fishers and reserve managers can approach reserve implementation with newfound confidence. People contemplating reserve creation can take heart from the experiences of others who are further along in the process. Their experiences prove that reserves can work successfully across a wide range of ecological and socio-economic conditions.

We highlight the factors that govern reserve success using studies that encompass a diverse range of habitats, and a size spectrum ranging from very small to very large, which and spans artisanal, recreational, small-scale, and industrial fisheries.

3. Evidence for build-up of spawning stocks in marine reserves

Increases in spawning stocks of exploited species are the most frequently reported effects of reserves and have been reviewed in depth many times (see Introduction). An analysis by Halpern (in press), reviewed effects of 89 studies of reserves that were at least partially closed to fishing, revealing that 63% of reserves increased the abundance of protected animals, 80% increased their average size, 90% increased biomass, and 59% increased diversity (number of species per unit of census area). On average, reserves doubled abundance, biomass increased nearly three times, protected animals were a third larger and diversity increased by a third. The magnitude of gains was independent of reserve size. In terms of these aggregate measures, small reserves performed as well as the largest.

Halpern's (in press) study has important limitations that indicate his figures underestimate the long-term performance of well-protected reserves. For statistical reasons he aggregated across species but this obscures some of the most striking responses to protection. Furthermore, the reserves differed in efficacy of their management and poorly enforced reserves will pull down aggregated performance measures. For example, Paddack and Estes (2000) found densities of kelp forest fish were only 12-35% higher in several small Californian reserves, which had a poor history of enforcement (Murray 1998). Halpern (in press) also lumped areas protected from some, but not all fishing together with fully protected areas. The timescale of reserve protection was also generally short in his sample, with most reserves benefiting from less than five years of protection.

A second line of reasoning also points to the changes reported by Halpern being underestimates of the potential of well-respected reserves. Studies contrasting areas subject to different intensities of fishing often show large differences in measures of abundance, biomass or body size of exploited species. For example, Hawkins and Roberts (2000) reported order of magnitude differences in biomass of predatory reef fish among lightly and heavily fished islands of the Caribbean. More than 54% of the biomass of fish assemblages in the remote and lightly fished northwestern Hawaiian Islands consisted of apex predators, compared to less than 3% in the heavily exploited main Hawaiian Islands (Friedlander and DeMartini 2002).

We can gain a better idea of the potential performance of reserves by looking at long-term studies of well-managed reserves where enforcement is good and/or compliance high. Such reserves often show order of magnitude increases in abundance, biomass or reproductive potential of prime fishery species. For example, the Leigh Marine Reserve in New Zealand has been protected from fishing since 1977 (Walls 1998). By the late 1990s, densities of fishable size individuals of an exploited bream, *Pagrus auratus*, had reached 5.8 – 8.7 times higher in the reserve compared to fished areas nearby (Babcock et al. 1999). The Leigh reserve also fostered increase in biomass and size of spiny lobsters, *Jasus edwardsii*, and they reached densities approximately 5 times higher than in fished areas (Kelly et al. 2000). A broader analysis of New Zealand reserves concluded that, in sites greater than 10m deep, they had supported annual rates of growth in spiny lobster biomass of 10.9%, in abundance of 9.5%, and in egg production of 9.1% (Kelly et al. 2000).

Russ and Alcala (1996a) reported that densities of large, predatory reef fish had increased 7-fold in 11 years of protection in the Apo Island reserve in the Philippines. South Africa has some of the world's longest established marine reserves. In the Tsitsikamma reserve (established 1964) underwater censuses of shore fishes suggested densities in the protected area were 42 times higher for the sparid fish *Chrysoblephus laticeps* compared to nearby fishing grounds (Buxton and Smale 1989). Catch-per-unit-effort (CPUE) for one sparid fish, *Petrus rupestris*, was 3000 times greater in the reserve compared to fished areas (Brouwer 2000), although this extreme difference may be a combination of a real density difference and a greater proportion of naïve fish in the reserve that were willing to take hooks (Davidson 2001).

In the Florida Keys, densities of yellowtail snapper, *Ocyurus chrysurus*, increased by more than 15 times in the fully protected Sanctuary Preservation Areas between 1997 and 2001 (Bohnsack 2002), and we have found similarly rapid responses and magnitudes of change in St. Lucian reserves (Gell et al. in prep. a).

Francour (1991) found that in rocky areas of the Scandola Nature Reserve in Corsica, mean biomass and densities of 11 species of fish studied were five times higher at no-take sites than fished sites after 13 years of protection. In seagrass areas some species were more abundant but the differences between the no-take and fished sites were not as great.

In the Cerbère/Banyuls-sur-Mer Marine Reserve in France, where spearfishing is banned, the densities of two important spearfishing target species, the moronid *Dicentrarchus labrax* and the bream *Sparus aurata* were significantly more abundant inside than outside the reserve (Jouvenel and Pollard 2001). *D. labrax* was nearly six times more abundant and had a mean length nearly 80% greater in the protected area than in the unprotected area (21.2cm vs 38.1cm). *S. aurata* were 13.6 times more abundant in the protected areas but there

was no significant difference in the mean length of this species. The authors attribute these differences to protection but note the possibility of behavioural differences of the fish between protected and unprotected areas.

Goñi et al. (2001a) found that the experimental CPUE for lobsters inside the Columbretes Island Marine Reserve in Spain were between 6 and 58 times higher from sites in the marine reserve than in fished sites.

In Tasmania, Edgar and Barrett (1999) found that the effectiveness of three marine reserves corresponded with the size of the reserve. The most effective was the Maria Island Reserve in which large fish became more than three times more common after 6 years of protection. The density of one species, the bastard trumpeter (*Latridopsis forsteri*), increased by two orders of magnitude over 6 years inside reserves. Outside reserves the species wasn't recorded at all.

Measures of increased abundance or biomass in reserves generally under-represent gains in reproductive output, because animals in reserves are larger than in fished areas, and large females produce more eggs per unit body weight than small (PDT 1990). For example, after five years of protection from fishing, 35% of blue cod, *Parapercis colias*, in the Long Island-Kokomohua Marine Reserve in New Zealand were >33cm long, compared to <1% of blue cod in nearby areas open to recreational fishing (Davidson 2001). In the Edmunds Underwater Park, in Washington State, USA, egg production by lingcod, *Ophiodon elongatus*, was estimated at 20 times greater than in surrounding fished areas, and that of copper rockfish, *Sebastes caurinus*, 100 times greater (Palsson and Pacunski 1995). Although just 10.1ha in size, the Edmunds park has been well protected since 1970. In the Tonga Island Marine Reserve in New Zealand, densities of large male spiny lobsters, *Jasus edwardsii*, reached 10 times greater in the reserve compared to fishing grounds within 5 years of protection, and egg output from reserves was estimated to be 9 times greater (Davidson et al. 2002).

It is clear that well protected reserves have the potential to rebuild stocks of exploited species to high levels. Long-term studies of such reserves give a different view from that suggested by Halpern's (in press) review. They show that biomass and reproductive output can increase to many times that in unprotected areas and gains of an order of magnitude are possible for some species. The size of effects that Halpern (in press) estimated are best considered as minimum expectations for short-term reserve performance.

4. Rate and timescale of build-up of spawning stocks

Halpern and Warner (2002) reviewed rates of response of protected species to reserve protection, using a subset of 80 reserves from the sample used by Halpern (in press). They found that, averaged across all reserves, measures of population density, biomass, organism body size, and species diversity responded quickly to protection. Average long-term mean values were reached within 1-3 years after protection and then remained stable for long periods. However, the strength of this type of meta-analysis is again reduced by the heterogeneity of the sample, mixing as it does well managed with poorly managed reserves, and partial protection with full.

Other reviews have also concluded that rates of population build-up are usually rapid, especially in the early years following protection (e.g. Roberts and Hawkins 2000, NRC 2001). Halpern and Warner's (2002) findings are supported by the many studies of reserves that have detected increases in protected populations within 1-3 years of closure to fishing (e.g. Roberts et al. 2001a, Bohnsack 2002). Aggregate biomass and abundance often double or triple within 3-5 years of protection. However, studies of well-managed reserves where there are reasonably long time-series of data post-protection, lead us to a different conclusion from Halpern and Warner (2002) regarding how long increases are sustained for.

Stocks of five families of exploited reef fishes quadrupled in biomass in reserves within six years of protection in St. Lucia (Roberts et al. 2001a, Gell et al. in prep. a). Apart from a dip in the rate of increase following a hurricane, there was no indication that biomass had peaked after this time or that rate of build-up was falling off. Russ and Alcala (1996a), reported a continuous linear increase in densities of large predatory fish in the Philippine Apo Island reserve over an 11 year period, and this trend has continued since then (G. Russ pers. comm.). Kelly et al.'s (2000) data on sustained rates of increase of spiny lobsters in New Zealand reserves, reported above, included reserves up to 21 years old. It is evident that there are rapid increases in many species, but gains continue to be made well beyond the first three years of protection.

The Merritt Island National Wildlife Refuge in Florida provides a good example of how reserve effects can develop over long periods. This area was closed to fishing in 1962 when it became the security zone for the Kennedy Space Center at Cape Canaveral (Johnson et al. 1999). It began supplying world record size fish to the surrounding recreational fishery after around 9 years of protection from fishing (Roberts et al. 2001a), and records continue to be caught there because of increasing numbers of exceptionally large fish that have benefited from long-term reserve protection.

Of course, populations in reserves will not increase indefinitely. Resources will ultimately become limiting, and there may be cascading effects of one species upon another, meaning that in some cases, what goes up may come down again. For example, populations of sea urchins in New Zealand and Tasmanian reserves have decreased as populations of their spiny lobster and fish predators have grown (Babcock et al. 1999, Edgar and Barrett 1999, Shears and Babcock 2002). Similar effects have been observed in Californian reserves

(Lafferty and Kushner 2000). Aggregate measures of change in biomass conceal such effects. Species with high population turnover rates will tend to respond to protection quickly, but their populations may fall as those of larger bodied predator species begin to recover (Pinnegar et al. 2000). A classic example is the transformation of rocky shores in Chile following protection. As the exploited predatory gastropod *Concholepas concholepas* recovered, it reduced densities of mussels and herbivorous gastropods turning bare rocky shores into beds of seaweed (Castilla 1999). We are only just beginning to see evidence of such changes in dominance of species in the longest established reserves.

5. Which species benefit from reserve protection?

For a species to benefit from protection in a reserve, it is necessary for some fraction of the population to spend a significant part of their lives there. This pre-condition has led to a widespread misconception that reserves will only benefit sedentary species, and the corollary that they will not benefit migratory species. A recent report produced for the FishAmerica Foundation, a recreational fishing industry lobby group, takes this as the starting point for a cursory analysis asking which species of fish will benefit from reserves (Shipp 2002). Shipp (2002) considered a series of species in US waters and then made subjective judgments on whether or not he believed they would benefit from marine reserves. In the great majority of cases he concluded reserves would offer no fishery benefit, but offered very little empirical evidence in support of his conclusions, citing only two papers that report studies of real marine reserves.

Certainly, some of the species that benefit most quickly from reserves are relatively sedentary, such as scallops (Murawski et al. 2000), abalones (Wallace 1999), and some fish (Russ and Alcala 1998, Murawski et al. 2000, Roberts et al. 2001a). However, populations of many species with greater mobility do build up in reserves. Indeed we are learning much about the behavioural complexities of exploited species by studying protected populations. For example, twenty years ago, few people thought that small reserves could protect populations of lobsters because in many species there are seasonal onshore-offshore movements that would take lobsters into fishing grounds. However, there is now compelling evidence from several different species in widely separated parts of the world that lobster stocks do build-up in reserves: from the Mediterranean (Goñi et al. 2001a,b), New Zealand (Kelly et al. 2000, Davidson et al. 2002), Australia (Edgar and Barrett 1999), Bahamas (Lipcius et al. 2001) and Canada (Rowe 2001, 2002), to name a few. It is now evident that a proportion of the lobster population shows high levels of site fidelity in each place studied (Kelly 2001, Rowe 2001).

The New Zealand sparid fish, *Pagrus auratus*, was another species not expected to benefit much from small, nearshore reserves because individuals seasonally move onshore-offshore. Yet they showed swift responses to reserve protection, again because a fraction of the population demonstrated strong site fidelity, while others moved (Willis et al. 2001).

South African research shows a similar pattern of differential propensity to move within a species. In a tag and recapture study, Griffiths and Wilke (2002) found that 7.4% of recaptured *Petrus rupestris* had moved distances between 200km and 1000km. The remainder of the population had much more limited movements, with 95% of recaptures occurring within a radius of 14km. Attwood (2002) reports similar behaviour in the Galjoen, *Dichistius capensis*, in which the furthest displacement of a tagged fish was 1300km, corresponding to almost the entire geographic range of the species. Nonetheless, many individuals have more limited ranges of movement and there have been marked increases in abundance and size of this fish in South African reserves (Buxton and Smale 1989, Attwood 2002). For four other shorefish species tagged by Griffiths and Wilke (2002), 95% of recaptures were within radii of 7km, 13km, 15km and 49km and they conclude that a minimum reserve length of 45km would be effective at protecting this multispecies complex. For reserves to work well, they must be scaled to the movements of the species protected or must be sited in places where they can offer transient protection to them. The latter effect is described below.

Fixed location marine reserves can also protect migratory species. Fishery managers already use closed areas to protect juvenile nursery areas of species that undergo ontogenetic migrations (i.e. move from one habitat to another as they grow). Such areas are a simple way to increase yields by preventing premature capture of young animals (Horwood et al. 1998). For example, juvenile lobsters are protected in Florida Bay until they are large enough to migrate to reefs in the Florida Keys where they can be captured (Davis and Dodrill 1980). There are large areas closed to trawling in Alaska and Kamchatka to protect juvenile Red King Crab (Armstrong et al. 1993).

The above examples point to ways in which reserves can benefit species that undergo annual migrations. The key is offering protection at places and times when the species are particularly vulnerable to capture. Roberts and Sargant (in press) point out that many migratory species aggregate or pass through migration bottlenecks where they can become highly vulnerable to targeted fisheries. For example, cod, *Gadus morhua*, populations in Newfoundland were intensively fished in nearshore spawning aggregation sites prior to the stock collapse in 1992 (Kulka et al. 1995). Spawning aggregations of Nassau groupers, *Epinephelus striatus*, and other species of snappers and grouper are being eliminated by over-fishing throughout the Caribbean

(Sadovy 1993, Heyman et al. 2001, Sala et al. 2001). Reserves that protect aggregation sites could significantly reduce overall fishing mortality. In the U.S. Virgin Islands, protection of a spawning aggregation site for Red Hind grouper, *Epinephelus guttatus*, has led to swift increases in average fish size and an increase in the availability of males in this hermaphroditic species (Beets and Friedlander 1999), despite covering just 1.5% of the fishing grounds (Bohnsack 2000). There is new evidence that species like cod may home to specific coastal spawning sites and would benefit from reserve protection in a similar way (Begg and Marteinsdottir 2000, J.A. Hutchings, pers. comm.). Even highly migratory species, like sharks, tuna and billfish, could benefit from reserves targeted to places where they are highly vulnerable, such as nursery grounds, spawning sites or aggregation sites like seamounts (Norse et al. in press).

Reserves could also be used to protect migration routes, such as that for female blue crabs, *Callinectes sapidus*, in Chesapeake Bay in the USA (Lipcius et al. 2001a). At present only their spawning area at the mouth of the bay is protected from fishing. Lipcius et al. (2001b) conclude that while this closure protects 11-22% of the spawning stock, this is below the level recommended for sustainable exploitation (28%). Lipcius et al. (2001a) propose an expansion of the sanctuary to include a deep-water migration route taken by females to reach spawning sites. This would have the advantage of protecting a sufficient fraction of the spawning stock while the fishery could target mainly males and younger crabs in shallower water.

Shipp's (2002) analysis fails to take into account any effects of habitat protection in reserves on the behaviour of species. There is a rich literature documenting how animal movement, social, reproductive and foraging behaviours are all influenced by habitat characteristics and food availability (e.g. Stephens and Krebs 1986, Krebs and Davies 1993). Animals respond to their surroundings and behaviours are not immutable over time. As habitats change and populations respond to protection, behavioural changes are inevitable. For example, following protection of the Leigh Marine Reserve in New Zealand, a fish species previously thought to be solitary began adopting haremic behaviour as their densities increased (Ballantine 1991). Improvements in habitat in reserves, such as increasing structural complexity and prey densities in areas previously trawled (Bradshaw et al. 2001, 2002), will likely increase growth and survival of animals present there (Lindholm et al. 1999). Reserves may also change previous behaviour patterns; for example, migratory species may spend longer in protected areas to take advantage of enhanced feeding conditions. Although not a migratory species, the coelacanth provides a possible example of just such an effect. In the St. Lucia marine reserve in South Africa, coelacanths occur in depths of 80-100m, much shallower than in other places. There is speculation that high prey densities in the reserve allow the fish to forage shallower than in places that are heavily exploited by artisanal fisheries, like the Comoros Islands (Plante et al. 1998, C. Attwood pers. comm.). As the number of long-term studies of reserves grows, we can expect to see more evidence showing that fishing not only reduces species densities, but also alters their behaviour.

Real reserves and fishery closures have already demonstrated benefits for fishery species as diverse as shellfish (Wallace 1999, Murawski et al. 2000), squid (Sauer 1995), crustaceans (Kelly et al. 2000, Rowe 2002), ascidians (Castilla 1999), and fish of a wide range of mobilities (Willis et al. 2001, Wantiez et al. 1997, Holland et al. 1993, 1996, Murawksi et al. 2000). Few reserves have yet been designed to benefit highly migratory species, but such animals could benefit from protection (Norse et al. in press). Models suggest reserves targeted to places of high stock vulnerability can protect spawning stocks and even enhance catches (Apostolaki et al. 2002, Roberts and Sargant in press). Contrary to claims by some critics (Shipp 2002), we conclude there are reserve designs that will offer some benefit to almost all species, with widely ranging species needing supplementary management outside reserves.

6. What is the evidence that fishery catches benefit from reserves?

It is evident from the foregoing account that the precondition for fishery benefits has been met: spawning stocks do build up in reserves. But do these gains translate into greater catches? Studies of fishery closures provided early evidence that year-round protection, or even seasonal protection, could increase yields. For example, in Japan, a 13.7km^2 area was closed to all fishing to protect Zuwai (Tanner) crab (*Chionoecetes opilio*) in 1983 (Yamasaki and Kuwahara 1990). Fisheries had been in decline for this species, partly due to by-catch mortality of juveniles in the unselective seine fishery, and in fish trawls. The closure was located at an average depth of 270m in an area where female crabs aggregated to mate each August to November. Over four years the closure increased the fraction of large male crabs in the area from 10 to 42%. CPUE data from experimental fishing in the closed area indicated an increase in crab densities over time. By the end of the study, seine vessels had begun concentrating fishing effort around the refuge and boats close to the refuge generally achieved higher catches of large male crabs (the fishery is entirely for males). Tagging showed that crab movements regularly took individuals from the refuge into fishing grounds, thereby supplying the fishery through spillover (Yamasaki and Kuwahara 1990).

Unfortunately, there have been few well-studied cases like that of the Zuwai crab, and until recently, the best evidence that reserves enhance catches in surrounding fisheries has been from more localized studies

documenting increased catch per unit effort (CPUE) close to reserve boundaries. For example, McClanahan and Kaunda-Arara (1996) and McClanahan and Mangi (2000) showed that trap fishers close to the Mombasa Marine National Park in Kenya enjoy three times greater catch per trap than those further away, despite higher trap densities close to the reserve. These fishing spots are so lucrative they are now reserved by informal agreement for the most senior fishers (Rodwell 2001).

Changes in fishing patterns, like that of the Mombasa fishers, are often the first indication that reserves are beginning to work, although the evidence is often anecdotal. For example, lobster traps ring the boundaries of the Leigh Marine Reserve in New Zealand (W. Ballantine pers. comm.), the Bicheno reserve in Tasmania (N. Barrett pers. comm.), and the Anacapa Reserve in the California Channel Islands (R. Fujita pers. comm.). Ramos-Espla and McNeill (1994) reported that catches of key species around the Isla Tabarca Marine Reserve in Spain were up by 50-85% over pre-protection levels, although they did not provide supporting data in this paper. In the Great Barrier Reef Marine Park in Australia, the Shelbourne Bay Cross Shelf Transect was closed to fishing in 1988. In 1990 trawl fishermen interviewed about the closure reported increased catches through 'fishing-the-line', as fishing close to reserve boundaries is called. Compliance with the closure was reported to be about 80% and some of the fishers interviewed were in favour of permanent closures (Shorthouse 1990). However, quantitative evidence of fishing-the-line is beginning to be collected. For example, Bohnsack (2002) found lobster traps set preferentially close to boundaries of the Sambos Ecological Reserve in the Florida Keys. Satellite transponders on boats in the New England scallop fishery have shown vessel tracks clustered close to the borders of large areas closed to groundfish trawling (Murawski et al. 2000).

Fishers would presumably not waste effort fishing close to reserve boundaries if they were not rewarded with better catches but few studies have yet quantified spillover catches. Kelly et al. (2002) studied catches from one spiny lobster fisherman fishing around the Leigh Marine Reserve in New Zealand. His catches were more variable around the borders of the reserve than those in areas further away, but large catches were more frequent. On average, catches from close to the reserve were similar to landings from other sites, despite the shallow habitat near the reserve being generally less productive for lobsters, suggesting the reserve was supplying this fishery.

Several studies have recently confirmed predictions that marine reserves can increase overall catches. In St. Lucia, Roberts et al. (2001a) found that in five years, CPUE of large fish traps increased by 46% and small traps by 90% in fishing grounds around a network of four reserves in the Soufriere Marine Management Area. Since numbers of fishers and overall fishing effort did not change significantly over the course of the study, this suggests a greater total yield, despite contraction of fishing grounds to two-thirds of their former extent. A very similar effect of reserves has been seen in Egypt around a network of small reserves near the settlement of Nabq in the southern Gulf of Aqaba. There, CPUE of the trammel net fishery has increased by 66% in five years of protection, from 0.79 kg per net per hour to 1.31 kg per net per hour. Underwater visual censuses showed corresponding increases in stocks of target fish in the reserves (Galal et al. 2002). Interestingly, the Nabq area was relatively lightly fished compared to many places where reserves have been established (Roberts and Polunin 1992). These reserves have promoted build-up of populations of larger predatory species, including groupers (Serranidae), emperors (Lethrinidae) and snappers (Lutjanidae). Such fish are often rare and may respond slowly to protection in places that have been intensively exploited (Roberts 2000b).

In the Philippines, CPUE of the hook and line fishery around the 0.74 square kilometre Apo Island Reserve increased ten-fold over twenty years of protection, from 0.13-0.17 kg per person per hour in 1980/1 to 1-2 kg per person per hour in 1997-2001 (Maypa et al. 2002). Total yield of the Apo Island fishery is among the highest in the Philippines and has been stable for the last 15 years at 15-30 tonnes per square kilometre per year. (Although CPUE increased, total yield did not due to reducing effort in other associated fisheries, such as drift gill nets (Maypa et al. 2002).)

Experience from Georges Bank in the Gulf of Maine shows that fishery closures can work at much larger scales. In 1994, in an effort to stem chronic declines in groundfish stocks, three large areas totaling 17,000 square kilometres were closed to all fishing gears that target these species, including gears like scallop dredges that might catch them incidentally or damage their habitat. Murawski et al. (2000) judged the closed areas a major success. They significantly reduced fishing mortality of depleted groundfish stocks, with clear increases in abundance and size of haddock and flounder. Georges Bank cod stocks are also creeping up, in part because of the closures, but also due to a package of gear and effort restrictions that complement them (NEFSC 2001). These improvements are beginning to have an impact on fish catches. At a 2001 meeting of fishers and scientists in Oregon, a Cape Cod fisherman described how prior to the closures, he had to steam 70 miles one way for a day's catch of around 270 kg of cod, but these days he travels just 30 miles to the edge of a closed area and catches 500 kg.

Groundfish gains aside, the most obvious benefit has been the dramatic recovery of the fishery for scallops, *Placopecten magellanicus*. After five years of protection their populations rebounded to 9 to 14 times the density of legal size scallops in fished areas (Murawski et al. 2000), and catches in surrounding areas increased. The closures have rehabilitated one of the most valuable fisheries in New England.

6.1 What are the mechanisms involved in delivery of fishery benefits?

Of the two mechanisms for transfer of animals from reserves to fisheries – larval export, and spillover of adults and juveniles – there is most evidence for spillover. Spillover can take place through several mechanisms: (1) random movements of animals taking them across reserve boundaries, (2) density-dependent movements taking animals from high density populations in reserves, to lower density fishing grounds, (3) directed movements, such as daily or seasonal migrations, and (4) ontogenetic habitat shifts which take animals out of reserves into different habitats. Sometimes it is hard to separate these processes, especially mechanisms 1-3, but other cases are clear. For example, fishery closures of nursery grounds take advantage of ontogenetic habitat shifts. In the Cyprus trawl fishery, areas close to shore were closed to fishing to protect juvenile fish that later moved offshore and into the trawl fishery as they grew (Garcia and Demetropoulos 1986).

Tagging data and the development of fishing-the-line supported the view that spillover of Zuwai crab was occurring from the fishery closure in the Sea of Japan (Yamasaki and Kuwahara 1990). Similarly, tagging and tracking of protected lobster populations in Newfoundland and New Zealand reserves indicates movement from reserve to fishery (Kelly 2001, Rowe 2001), with a pronounced seasonal component as lobsters move between nearshore and offshore habitats. Reserves in New Zealand also export the bream *Pagrus auratus* on a seasonal basis as fish move from shallow to deeper areas (Willis et al. 2001). In East Africa, catch rates of fishers around the Mombasa Marine National Park were clearly augmented by fish moving from the reserve (McClanahan and Mangi 2000). Reef fish frequently undergo daily foraging migrations, particularly from resting sites on reefs to nighttime foraging areas over sandy or seagrass habitats (e.g. Meyer et al. 1983). Such movements commonly span 100s of metres to a kilometre and can take fish in and out of reserves (Holland et al. 1993, 1996).

In the Philippines, Russ and Alcala (1996b) present evidence of spillover from the Apo Island Reserve based on annual fish counts. Fish densities began to increase in fishing grounds close to the reserve boundary after 9 years of protection from fishing. However, spillover had probably been occurring for much longer. It is only after the rate of spillover exceeds the rate of its removal by fishers that fish populations will build up close to reserves.

Catches of world-record sized fish around the Merritt Island National Wildlife Refuge in Florida are undoubtedly caused by spillover of large, old fish (Johnson et al. 1999, Roberts et al. 2001a). Tagging studies at the reserve also show export of smaller gamefish to the nearby recreational fishery (Stevens and Sulak 2001). Forty years of protection should be long enough for population densities in the reserves to have reached natural levels and it may be that spillover is now density-dependent, although studies have not yet addressed this question. Lizaso et al. (2000) note that while density-dependent spillover is frequently assumed, there has been little research to test whether this is the case. Attwood (2002) analysed detailed tagging data for galjoen in South Africa and found that movement rates from reserves to fishing grounds were independent of fish densities in reserves. Here spillover is principally via both random and directed movements of fish that are probably in search of higher quality habitats (Attwood 2002).

Not all marine reserves have led to spillover. Davidson (2001) conducted experimental fishing to measure effects of the Long Island-Kokomohua Marine Reserve on populations of blue cod. While catch rates in the reserve rose by 300% over the seven year study, they remained constant in control sites located 1.5–5.6km from the reserve boundary. It may be that densities had not yet built up high enough to trigger movement and spillover from the reserve, or that the species is so sedentary, that any fishery enhancement will depend on larval export rather than spillover.

There is far less evidence as yet for larval export from reserves to fishing grounds. This is not because larval export is rare, but because of the great difficulty of determining the sources of animals recruiting to a fishery (Palumbi 2000). Wherever spawning stocks increase in reserves above those in fishing grounds, then area for area, reserves can be expected to outperform fishing grounds in their contribution to future recruitment. That contribution will depend on the scaling of reproductive output from reserves compared to fishing grounds. For example, the Edmunds Underwater Park in Washington State covers just 550m of coastline, but with 100 times greater reproductive output, copper rockfish in the reserve could supply eggs equivalent to production from 55km of exploited shoreline in Puget Sound (Palsson and Pacunski 1995). Reserve networks at Nabq, Egypt, and in St. Lucia were designed to promote spillover, with small reserve units interspersed with fishing grounds along the coast (Roberts et al. 2001a, Galal et al. 2002). However, the magnitudes of improvements in stocks outside reserves in these locations, and also at Apo Island (Maypa et al. 2002) make it likely that fisheries are being supplied by significant larval export too.

The easiest situations in which to detect larval export are those where stocks have been severely depleted prior to reserve establishment. At an experimental scale, Tegner (1993) increased recruitment of juvenile green abalone, *Haliotis fulgens*, by transplanting adult broodstock to a location on the Californian coast where they were rare. In Chile a 3 year closure of the squat lobster (*Pleuroncodes monodon*) fishery ending in 1991 led to increases in biomass exceeding all previous estimates since the population was first monitored in

1979. It also led to the re-expansion of the distribution of the species by more than 50km into areas where it had previously been fished out (Roa and Bahamonde 1993). This expansion was probably driven by larval dispersal.

In Fiji, the Ucunivanua community established a 24ha area of mudflat and seagrass habitat in front of their village and closed it to fishing for clams, *Anadara* sp., in 1997 (Tawake et al. 2001, see Fiji case study). Four years later, clam abundance and size had increased dramatically in the closed area, with densities 14 times greater than at the time of protection. Densities also increased by more than 5 times in adjacent fished areas, largely through juvenile recruitment. These increases suggest export of larval clams produced by protected populations.

The best demonstration to date of catch enhancement through larval export comes from fishery closures on Georges Bank. The rebound of scallop stocks in closed areas provided a clearly identifiable source of larvae to supply surrounding fisheries. While scallops can move by 'clapping' around, they are unlikely to get far this way. Instead, larvae exported from the closed areas supplied downstream fishing grounds. Oceanographic models that simulate larval dispersal trajectories (Lewis et al. 2001), predict recruitment plumes outside reserves that coincide with areas where satellite data show fishing effort was concentrated (Rago and McSherry 2002).

6.2 Are reserves beneficial only to some kinds of fisheries?

Reserves are often claimed to be only appropriate to particular kinds of fisheries (usually artisanal), particular habitats (usually coral reefs), or to sedentary species. However, the range of examples described above shows that this is not the case. Reserves and fishery closures work for a wide range of species possessing a wide spectrum of mobilities. They have performed well across a broad variety of habitats including coral reefs, temperate rocky reefs, estuaries, seagrass beds, and temperate continental shelves. They have also supported many different kinds of fishery, from the artisanal fisheries of coral reefs, through small-scale nearshore fisheries for animals like lobsters, to large-scale industrial fisheries for species like scallop, haddock and flounder.

7. How large do reserves have to be for fisheries to benefit?

There are two facets of reserve size to consider: the area of individual reserves and the fraction of fishing grounds to be protected. First we look at the size of individual reserves. Several recent papers have considered size and spacing of reserves (e.g. Kramer and Chapman 1999, Roberts and Hawkins 2000, Roberts et al. 2001b, Botsford et al. 2001). For reserves to promote build up of exploited species, they need to be large enough to protect animals from fishing by encompassing their ranges of movement. The more wide ranging species are, the larger reserves need to be to offer them protection (but see earlier discussion of reserve designs that will benefit migratory species). If fisheries are to benefit from spillover, then reserves must be small enough so as not to staunch spillover. This is because edge-to-area ratio of a reserve falls as its size increases and so boundaries of large reserves will be less porous than small (Kramer and Chapman 1999).

Halpern (in press) has shown that reserves work well across a wide range of sizes, from a few hectares to hundreds of square kilometers, provided they are well respected. Based on consideration of the range of movements of animals important to fisheries, and dispersal distances of their offspring (Grantham et al. in press, Palumbi in press, Shanks et al. in press), it is evident that we need to include a range of reserve sizes in networks. Roberts et al. (2001b) argue that such networks in coastal areas should contain reserves of a few to a few tens of kilometers across, separated by distances of a few to a few tens of kilometers.

Now we turn to the question of how much of the fishing grounds should be protected. Both small and large reserves can produce local catch enhancements close to their boundaries. Examples of fishing-the-line can be found around small reserves like Leigh in New Zealand (5.2km^2)(Kelly et al. 2002), and even the tiny Anse Chastanet reserve in St. Lucia (2.6ha)(Roberts and Hawkins 2000). Most existing marine reserves are much too small to produce overall catch enhancements. The 13.7km^2 Zuwai crab reserve in Japan comprised only 2% of the fishing grounds for this species and so probably had little effect on total landings, which held steady over the course of Yamasaki and Kuwahara's (1990) five year study. Regional benefits require reserves to be scaled up. There is now a large body of theoretical work offering insights into what fraction of an exploited stock needs protection to yield fishery benefits (reviewed in Roberts and Hawkins 2000, NRC 2001). In brief, reserves that cover between 10 and 50% of the sea generally produce maximum benefit to fisheries, in terms of catch. The majority of studies suggest that protecting 20-40% of fishing grounds will produce maximum benefits to fisheries. In any particular case, the optimal fraction depends on the species' mobility (more mobile species generally need more reserve), its vulnerability (more vulnerable species need more reserve), the quality of management outside the reserve (well-managed areas need less reserve), and the intensity of fishing (more intensive fisheries need larger reserves to support them). Pezzey et al. (2000) created a bioeconomic model of reserve effects on Caribbean reef fisheries, and tested it under situations of widely differing fishing intensities.

They concluded that reserves covering 21% of the fishing grounds would be needed in moderately fished Belize, 36% in heavily fished St. Lucia, and 40% in intensively fished Jamaica.

The most striking effects of reserves on fisheries come from situations where reserves have constituted a large fraction of the fishing grounds. For example, in St. Lucia, 35% of local reef habitats were protected within the Soufrière Marine Management Area (Roberts et al. 2001a), corresponding almost exactly with the optimal fraction predicted by Pezzey et al. (2000). In Nabq, reserves constitute approximately one third of former fishing grounds in the Nabq Managed Resource Protected Area, and there is a large area of reef to the south protected within the Ras Mohammed National Park which may also serve as a source for new recruitment (Galal et al. 2002). The 17,000km^2 closed to groundfish fishing and scalloping on Georges Bank comprises approximately 25% of the Bank. The Merritt Island National Wildlife Refuge in Florida constitutes approximately 22% of the estuarine habitat used by recreational fishers (Bohnsack 2000). Apo Island's reserve covers less area, just 10% of the fishing grounds (Maypa et al. 2002). However, its success is partly attributable to good management in adjacent fishing grounds where all destructive fishing methods have been banned. Similarly, although they have not yet been in place for long enough to have contributed to catches of world record sized fish, other fishery management measures recently introduced in Florida will complement the Merritt Island reserve in delivering benefits to recreational fisheries for red drum, black drum (*Pogonias cromis*) and spotted seatrout (*Cynoscion nebulosus*) (Wickstrom 2002).

Attwood (2002) estimated that the 9% of galjoen habitat in South African waters that was protected in reserves was large enough to have stabilized catches in adjacent heavily overexploited linefisheries but had not been large enough to increase catches. Sometimes, reserves may only be large enough to stem the rate of decline in a fishery rather than foster recovery. For example, the area around the 6.2km^2 Mombasa Marine National Park had the highest local catch rates (5.5kg.ha^{-1}.month^{-1}), despite having the highest density of fishers (McClanahan and Mangi 2001). However, over five years the catches there were still declining at a rate of 250g.day^{-1}, although this was at a lower rate than in fishing grounds further from the reserve which averaged losses in yield of 310-400g.day^{-1}.

The question is often raised as to whether we should start small or large in establishing reserves. If large and rapid fishery benefits are desired, then a large start is necessary. Large starts proved valuable in St. Lucia, Georges Bank and Nabq, some of the most successful demonstrations of fishery enhancement to date. A disadvantage of this approach is that the initial costs and disruption of fishing patterns borne by fishers are higher (Holland and Brazee 1996, Sladek Nowlis and Roberts 1999). Small starts, like the 2% closures for Newfoundland lobster (Rowe 2002), cannot possibly achieve more than minor local effects on catches. However, they can play a very important role in proving to communities that the concept works, and so build confidence among fishers for more ambitious reserves. In the Florida Keys, increased fish stocks in the small Sanctuary Preservation Areas helped persuade fishers to create the much larger Dry Tortugas Ecological reserve in 2001, which covers 150 square nautical miles (http://www.fknms.nos.noaa.gov/tortugas/). In Fiji, small no-take areas have proliferated into networks as people have adopted the approaches they have seen succeed in neighbouring communities (Tawake and Aalbersberg 2001, Tawake et al. 2001). This has also happened in the Philippines, and the community-motivated increases in the coverage of marine reserves seem to be an effective way to reach sustainable levels of protection.

8. What are the timescales of benefit to fisheries?

Studies in St. Lucia, Nabq, Apo Island and Georges Bank have all demonstrated catch enhancements in nearby fisheries within five years. Detailed data on catches from St. Lucia showed that CPUE for trap fisheries increased above levels at the time of reserve protection within 3-4 years (Gell et al. in prep. a, Soufrière Marine Management Area case study). In all cases, early catch improvements came from 'quick response' species, generally those with high rates of population turnover. In Georges Bank, scallops rebounded quickly following exclusion of bottom fishing (Murawski et al. 2000, see Georges Bank case study). There have also been improvements in stocks of haddock and flounders, and these are now beginning to spillover into fisheries (Paul Howard, New England Fishery Management Council, pers. comm.), but cod have been slower to respond. Likewise, Roberts et al. (2002) attribute catch enhancements around the Soufrière Marine Management Area in St. Lucia to 'high turnover' species like small snappers, squirrelfish and grunts. Populations of several larger species of snappers and grunts are building up in these reserves but have yet to become common in landings (Gell et al. in prep. b). It will take longer for benefits to become apparent for such 'slow response' species.

The Merritt Island National Wildlife Refuge in Florida elegantly demonstrates how species' life histories influence rates of response to reserve protection and shows how benefits can build over long periods (Johnson et al. 1999, Roberts et al. 2001a). The three species involved in the recreational fishery varied widely in terms of longevity. Spotted seatrout reach 15 years old, red drum live to 35 and black drum to 70. Only after fish had been protected for long enough could they reach sizes greater than existing world records and only then could new world records be set by captures of fish emigrating from the reserve. This point was reached 9 years

after the reserve was created for spotted seatrout, 27 years for red drum and 31 years for black drum (Roberts et al. 2001a). The greater a species' longevity, the longer it took to reach record sizes. The protracted timescales over which populations of older, larger and more fecund animals grow in reserves indicate that egg production from protected populations will increase over long periods as they gradually develop an extended age structure.

Habitat effects of reserves can also take a long time to develop, but again can continue to grow over long periods. For example, recovery from the impacts of trawling can require settlement and growth of a wide range of invertebrates as well as the re-establishment of natural processes of sediment movement and bioturbation (Auster 1998). After five years of protection there were clear indications of habitat improvements in areas closed to trawling on Georges Bank. They included large increases in the density, biomass, species richness, and production of bottom-living animals relative to areas open to trawling (NRC 2002, J. Collie pers. comm.). After five years of monitoring the closure, there was no sign that recovery was complete (J. Collie pers. comm.). In the Irish Sea, Bradshaw et al. (2001), found that a decade of protection from trawling led to the development of more heterogeneous and diverse bottom communities. Full recovery from trawling can take decades (Bradshaw et al. 2002, NRC 2002). It must also be remembered that recovery is contingent on the intensity, extent and duration of past disturbance and the proximity of undisturbed or less disturbed areas. Both will limit the availability of species to recolonize protected areas.

Where fishing has indirect effects, improvements in habitat may not begin to accrue until certain conditions have been met. In East Africa, fishing has well-known indirect effects on coral populations. Over-exploitation of the fish predators of herbivorous sea urchins has led to population outbreaks of urchins. At high densities urchins reduce coral cover as they scrape algae from the reef (Carreiro-Silva and McClanahan 2001). This in turn reduces habitat structural complexity which reduces the capacity of the reef to support fish production (Roberts 1996). Using a bioeconomic model, Rodwell et al. (in press) show how habitat improvements in East African marine reserves can enhance the magnitude of reserve contributions to surrounding fisheries. Similar benefits of increasing biomass of benthic organisms and improved structural complexity can be expected in other habitats, such as former trawling grounds like those on Georges Bank (Lindholm et al. 1999). Habitat improvements in reserves can improve conditions for recruitment and offspring survival. For example, small juvenile red hake, *Urophysis chuss*, shelter in the shells of scallops (Kramer et al. 1997) and so increased scallop populations in closed areas of Georges Bank will probably contribute to red hake recovery there. Rogers-Bennett and Pearse (2001) found improved survival of juvenile abalones under spine canopies of urchins protected from fishing in Californian reserves. However, because habitat improvements can take a decade or more to develop, such effects will rarely be observable in the earliest years of reserve establishment (Rodwell et al. in press).

Full protection from fishing increases the likelihood of quick fishery benefits over partial protection measures. This is because such closures are likely to encompass more 'quick response' species, and habitat recovery from fishing impacts can begin immediately and in full. Furthermore, full protection increases the likelihood of protecting species whose behaviour and reproductive characteristics promote spillover and export of offspring to surrounding fisheries.

An interesting point to note here is that while fishers often see that fish are building up in closed areas, they are sometimes less able to perceive or are unwilling to acknowledge gradual increases in their catches. Fishers may therefore believe that they are being excluded from the benefits of the closed area (e.g. SMMA, St Lucia, pers. obs., Misali Island Marine Conservation Area, Tanzania, Levine 2002). Consequently fishers often suggest that reserves be reopened for a short time to allow them to benefit from the increased number of fish. As examples from Tanga in Tanzania (Horrill et al. 2001), and Sumilon Island in the Philippines (Russ and Alcala 1999) show, reopening areas leads to rapid decreases in biomass and associated catch. The short-term benefits of fishing in a reopened area are very unlikely to match the long-term security in catches a reserve can provide if protection is maintained (Sladek Nowlis and Roberts 1999).

9. What are the spatial scales of benefit to fisheries?

The great majority of studies have looked for effects inside reserves and in exploited areas close to their boundaries. There is a practical reason for doing so – most existing reserves are too small to show measurable fishery effects at regional scales. Reserves tend to be studied locally where their effects will be easier to detect. What is encouraging about local scale research is that almost all the available evidence suggests reserves deliver benefits to fisheries nearby. This means local efforts to establish reserves will be rewarded by local benefits.

The spatial extent of spillover benefits will be constrained by the mobility of species leaving reserves, and the intensity of fishing close to reserve edges. In the case of coral reef fisheries, this restricts the majority of spillover to areas within a kilometer of reserves (Russ and Alcala 1996b). In St. Lucia, Apo Island and Nabq, the fishers who gave up their fishing grounds now obtain larger catches fishing near reserves. In Jamaica, Munro (2000) tagged reef fish in the Discovery Bay Fishery Reserve, and found that the majority were recaptured

within or near the reserve, although some were caught up to 26km away. A few species such as the mojarra, *Gerres cinereus*, and the grunt, *Haemulon aurolineatum*, were never captured outside the reserve.

The Merritt Island Wildlife Refuge has a greater sphere of influence. Movements of the recreational fish species can take them tens of kilometers from the reserve (Stevens and Sulak 2001), and this is reflected by the spatial distribution of sites in Florida where world record sized fish have been caught (Roberts et al. 2001a). Similar scales of movement are evident for recreational line fish species in South Africa (Attwood 2000).

Where reserves protect nursery grounds, spillover can take place at a larger scale as animals grow and move to new habitats. Reserves designed to offer refuge to migratory species at specific places where they are highly vulnerable to capture (Sala et al. 2001), provide regional fishery benefits. For example, protection of the small spawning aggregation site of red hind grouper in the U.S. Virgin Islands produced generalized increases in fish size (Beets and Friedlander 1999).

Where intensive fishing-the-line takes place, much of the spillover may be caught locally. This is sometimes seen as a problem but it is not necessarily a bad thing. The magnitude of spillover may be similar regardless of where it is caught, but it does mean fishers living further away could gain less. One of the potential costs of reserves to fishers is increased travel times and transport costs to reach legal fishing grounds. However, this problem is easily overcome by creating networks of reserves with each unit distributing local benefits. This approach has been tried in Chile where Castilla et al. (1998) recorded increased travel time, and decreased profit for fishers who did not have marine reserves (incorporated in Management and Exclusion Areas) compared to those who did. This is because the fishers without reserves had to travel further to collect sufficient invertebrates from populations that had been severely depleted by overfishing. Fishers from sites with reserves had CPUE levels 4-10 times higher than those obtained by fishers without.

Larvae exported by reserves could potentially spread reserve benefits much more widely, although the dispersal scale of a species will limit the extent of the recruitment plume from any reserve. In species which disperse short distances, larvae may settle close to reserves. For example, abalones tend to disperse short distances (Tegner 1993) and so to achieve regional benefits for these species, reserves must be patterned around local populations (Jamieson 2000, Shepherd and Rodda 2001). In South Africa, a limpet species recolonized shores denuded by catastrophic floods at a rate of 15 km per year (Dye et al. 1994). On the Georges Bank, the distribution of fishing effort for scallops suggests that these they have recruitment plumes of 50-70km (Rago and McSherry 2002). Bio-physical modelling of the movement of haddock eggs, larvae and early juveniles from the Emerald and Western bank on the central Scotian shelf in Canada has shown that though the early haddock stages usually remain in the vicinity of the bank on which they were spawned, there are years in which they drift downstream and recruit to an adjacent stock (Frank 1992, Frank and Brickman 2001).

Longer dispersal phases have the potential to spread fishery benefits more widely, and many species do spend weeks or months in the plankton (Shanks et al. in press). However, our understanding of the scales of larval dispersal is rapidly changing and it seems that many larvae, even of species with relatively long dispersal stages, can settle locally (Swearer et al. 1999, 2002, Warner et al. 2000, Warner and Cowan 2002, Barber et al. 2000, 2002). Nonetheless, there is considerable evidence for extensive transport (Yeung et al. 2000). As with spillover, networking reserves is the best way to ensure that benefits of larval export are spread across a region.

10. Can reserves prevent collapse of intensively exploited fish stocks?

Some people think that the criterion by which fishery benefit of reserves should be judged is whether or not catches in the area remaining open, have increased to levels above the total yielded by the original area of fishing grounds. Reserves in St. Lucia, in the Georges Bank scallop fishery, and probably also in Nabq do appear to have achieved greater overall yields than at the time of protection (Hart in NEFSC 2001, Roberts et al. 2001a, Galal et al. 2002). However, if pre-reserve catches are at unsustainable levels, this is not a fair performance metric, since the original landings trajectory is one of decline. Models predict that under such circumstances reserves of sufficient size should prevent collapse in yields at high fishing intensities, although at lower levels of total catch than from a theoretical optimally managed fishery without reserves. (Such 'optimal' management is almost never achievable in real fisheries due to the ecological, social and management complexity of exploited systems.) Where fishing intensities are difficult to control, for example in widely dispersed artisanal or recreational fisheries, this could be a very useful fishery benefit of having reserves.

In South Africa, Attwood (2002) found that outside reserves, fishing intensities for galjoen are at unsustainable levels, and using a model based on movement data, showed that reserves have prevented collapse of the fishery for this species by stabilizing catches at productive levels. Similar effects of South African reserves have been inferred for other species. Kyle et al. (1997) concluded that the intensive but stable rocky shore fishery for mussels (*Perna perna*) in Maputaland could be sustained at such intensities because of 115 km of adjacent unfished coast in marine reserves. Yields were much lower at sites in Transkei where there is very little marine reserve area.

An important fishery benefit of reserves is to allow long-lived species the opportunity to develop natural, extended age structures. This increases the fraction of larger, more fecund fish and provides population resilience in the face of environmental variability. Marine ecosystems are often characterized by regime shifts in which there are decadal shifts in environmental conditions (Steele 1998). Under such conditions, many species experience sporadic peaks in recruitment separated by lengthy intervals when conditions for offspring survival are poor. Reserves provide refuges for fish to persist across periods where conditions are poor for survival and recruitment of their offspring. Modern fisheries usually reduce populations to only a handful of reproductively active age-classes. For example, 10-20 year old cod made up 48% of total cod biomass on the Grand Banks of Newfoundland in 1962, 8% in 1990 and 0% thereafter (Longhurst 1998). This means that without reserves, stocks can collapse completely during periods when conditions are unfavourable for offspring survival, and so fail to recover when better conditions return. By contrast, populations of older fish in reserves can promote recovery as soon as conditions improve.

11. Are there cases where reserves have not worked?

The commonest reason for reserves or fishery closures failing to foster recovery of exploited species is that they protect the wrong kind of habitat. For example, concerns about impacts of bottom trawling on fisheries for red king crabs (*Paralithodes camtschaticus*) led to an emergency closure of an area in the eastern Bering Sea in 1987. Following the closure, Armstrong et al. (1993) failed to find any significant changes in abundance of female and pre-recruit male crabs in surveys inside and outside the refuge. They concluded the refuge was in the wrong place, having left out important juvenile habitat, plus breeding and hatching grounds.

Some species may fail to respond to protection if reserves are designed around the needs of other species. For example, Fernandez and Castilla (2000) showed that a marine reserve in Chile had not affected abundance of stone crab, *Homalaspis plana*, in contrast to marked benefits for other exploited benthic species. They found that for stone crabs the reserve protected neither their favoured conditions of wave exposure, nor the most suitable substrate. This and other reserves in Chile had been designed to benefit other commercial species like the snail *Concholepas concholepas*.

Another reason for failure is that low quality habitat was protected, perhaps as a result of fishers being unwilling to give up prime fishing sites. For example, Heslinga et al. (1984) examined the performance of a series of areas in Palau protected for over 20 years from fishing for *Trochus niloticus* molluscs. They found that on average, refuge sites had only half as many *Trochus* as nearby exploited areas. Reserves had failed because they were set up in areas of marginal habitat for the snail, where few people ever bothered to fish.

There are several other reasons why populations might fail to recover in reserves. Species that have been driven to very low population densities may suffer reproductive failure as a result. This is known as an Allee effect and helps explain why the Queen Conch *Strombus gigas*, fishery in the Florida Keys has shown little sign of recovery despite being closed to fishing since 1986. Stoner and Ray-Culp (2000) studied conch in the Bahamas where they are still common, and found them highly prone to reproductive failure at low densities. They discovered that mating never occurred below densities of 56 conch per hectare, where according to the authors, conch are never much farther than 16m apart. Allee effects can be critical for organisms that have limited mobility and they may need assistance to reach critical densities. For example, in Palau, islanders maintain traditional 'clam gardens' where large giant clams (*Tridacna gigas*) are brought close together and protected from fishing.

Population recovery in reserves is also contingent on what remains there. Although the St. Lucian reserve network led to overall fishery recovery (Roberts et al. 2001a), populations of large groupers (*Mycteroperca* spp.) have not recovered. These species had been virtually extirpated by years of severe overfishing and there were no source populations left to restock marine reserves. By contrast, two smaller grouper species have responded well to protection (C.M. Roberts and J.P. Hawkins, unpublished data). In general, the more overfished a place is to begin with, the smaller the pool of species available to lead recovery once a marine reserve has been created. The species that tend to rebound first in reserves are small, fast-growing, early reproducing species that have probably formed the mainstay of the struggling fishery prior to protection. Rarer species may recover over longer timescales, but it may be necessary to wait for environmental conditions that favour strong offspring survival to promote their recovery.

Hutchings (2000) reported a poor record of success for fishery closures to rebuild collapsed stocks. He looked for evidence of recovery in 90 stocks of fish following efforts to reduce fishing mortality and found that few species had rebounded. These failures can be put down to factors such as Allee effects or behavioural disruption. For example, population collapse of Norwegian herring, *Clupea harengus*, seems to have led to loss of traditional knowledge among herring of migration routes to spawning grounds, with subsequent failure to reestablish them when stocks increased again (Dragesund et al. 1980). Some species may need habitats to recover before they will. For example, the recovery of kelp forests in New Zealand and Tasmanian reserves may have paved the way for population increases by species that live in these habitats (Babcock et al. 1999, Edgar

and Barrett 1999). Another reason for the poor results highlighted by Hutchings (2000) is that fishery closures are less likely to work than fully protected marine reserves. This is because they are often subject to continuing impacts such as by-catch, and habitat disturbance or damage by fisheries that continue in the same area. For species closures where fishers continue catching other species in the same area, the temptation to take a few 'forbidden' species may sometimes undermine closure effectiveness. This was a problem in invertebrate gleaning fisheries in Fiji (Tawake and Albersberg 2001).

What these and other examples indicate is that it is important to act to protect species before they have declined too far. It is better to use reserves to help sustain production rather than relying on them as a rebuilding tool when things go wrong.

Finally, people are the key factor in whether or not reserves work. Without support and compliance from fishers nothing can happen. Fiske (1992) identified lack of involvement by artisanal fishers in efforts to establish the La Parguera National Marine Sanctuary in Puerto Rico as one of the main reasons why the proposal failed (although a reserve has since been established there). In California, USA, a design for a statewide network of marine reserves was abandoned in 2001 and a new process begun due to lack of input from fishers early on (MPA news, April 2002). In the Pacific, the Guam Territorial Seashore Park was selected and designed by the Government of Guam with very limited public consultation. The community wasn't involved in management decisions once the park was established and it suffers from poor funding and enforcement. Gilman (1997) calls it a "paper park" and suggests that only by involving user groups in management, decision-making and education will the park succeed. Conversely, in the Seaflower Biosphere Reserve, San Andres Archipelago, western Caribbean, efforts were made from the outset to fully involve all sectors of the local fishing community in consultation and planning. This has resulted in a high level of support (M. Howard pers. comm., Howard 2001, Howard et al. 2001). In a review of marine protected areas in the northwestern Mediterranean, Francour et al. (2001) noted that the better the initial consultations were with all user groups the fewer subsequent conflicts and management problems arose.

12. What advantages do reserves have over existing fishery management tools?

Some commentators ask the question, why use marine reserves when there are other fishery management tools available? Surely it is better, they argue, to use tried and tested methods rather than something new like reserves. Hastings and Botsford (1999) showed mathematically that marine reserves can be designed to achieve the same level of catches as traditional effort controls on fisheries. However, they point out that reserves also offer other advantages over traditional management measures. Stockhausen and Lipcius (2001) used a population model to compare effects of different reserve configurations (covering 20% of the total habitat) and effort reduction (by 20%) on lobsters in Exuma Sound, Bahamas. They found that various performance measures (mean catch, larval production and population growth rate) ranked strategies from best to worst in the order (1) single large reserve, (2) several small reserves, (3) reduced effort, and (4) no management.

Sladek Nowlis (2000), again taking a population modeling approach, showed that reserves performed better than temporary closures or minimum size limits on fish. Under the range of conditions modeled, they generally produced greater long-term catches with less restriction of fishing activities than would be necessary with other management measures.

Reserves offer many advantages over existing tools. They protect sensitive habitats from disturbance and damage by fishing gears such as trawls. Although closures to mobile fishing gears can achieve this, they lack the breadth of protection that reserves can afford. For example, reserves prevent by-catch mortality by preventing by-catch altogether. They eliminate ghost fishing by lost or discarded gear. They prevent impacts on populations by vessels high grading catches (discarding lower value species to load up with high value species). They foster development of natural, extended age structures in populations, something that effort reduction alone cannot do. Reserves may also offer greater protection against political interference in fishery management, since politicians setting catch levels may be less willing to reopen marine reserves than they are to overrule other kinds of conservation measure proposed by fishery scientists (Roberts 2000a).

Reserves also provide refuges for species that cannot persist in areas that continue to be fished. For example, the barndoor skate (*Raja laevis*) has nearly disappeared throughout its northeast Atlantic range due to by-catch in trawls targetting other species (Casey and Myers 1998). However, there has recently been evidence from experimental catches in closed areas of Georges Bank of a good recruitment of juvenile skate to the protected areas (S. Murawksi pers. comm.).

By protecting intact, undisturbed ecosystems, reserves also provide benchmarks against which the effects of fishing can be judged. Such information can improve fishery management, for example by providing better estimates of natural or fishing mortality rates. In California, protected populations in two reserves in the Channel Islands allowed a better evaluation of the dive fishery for warty sea cucumber (*Parasticopus parvimensis*)(Schroeter et al. 2001). They revealed that in 3-6 years, fishing reduced densities of sea cucumbers

by 33-83%. Analyses of conventional CPUE data showed no declines and even a significant increase over time at one island.

Ultimately, it is not a question of whether reserves are better than other management measures. Instead, we should ask how to combine these tools to greatest effect? At one extreme, some fisheries could be managed with reserves alone, while others may not need them at all. In between there is a very large number of species whose management would be improved by incorporating marine reserves into the management toolkit. At all times it must be remembered that reserves are multi-species tools. The objective is to design them so that there is a maximum aggregate benefit across the spectrum of species and fisheries.

13. What are the social effects of marine reserves on local communities?

Studies of ecological effects of marine reserves far exceed those of their effects on people (Beaumont 1997). While a growing body of work is looking at fisheries effects, much is concentrating on changes in yields and catch per unit effort, measures that reveal very little about the experiences of the people involved. Few studies have directly assessed the social and economic impacts of marine reserves on fishers and the wider local community, but those that do give valuable insights into issues of equitability and coping strategies in fishing communities (Malleret-King 2000). Nance et al. (1994), in their study of the social impact of a temporary seasonal closure of the shrimp fishery off Texas, pointed out the importance of this type of social research. They reported "surprise and dismay" amongst the scientific committee for management of the closure when their research revealed that after six years, more than 40% of fishers interviewed did not understand what was the purpose of the closure.

One problem with social studies is that social impacts and perceptions of communities are dynamic and change rapidly over time. The best way to build up a picture of social impacts is with long term socio-economic monitoring, but this has rarely been attempted. One of the best examples of formalised socio-economic monitoring is the work of Leeworthy (2001), who monitored the Sambos Ecological Reserve in the Florida Keys National Marine Sanctuary for four years (T. Murray and Associates 2001). One area of particular interest is the distribution of costs and benefits of the reserve across the community, and whether people who lost the most (those who previously fished in the closed area) also benefited most from the biological and economic consequences of protection. Contrary to prior expectations among fishers, he found that all fishers locally experienced increases in income over the study period. However, those displaced by the reserve gained an average of 67% compared to 22% gained by fishers further away.

Poaching, which occurs to some extent in virtually all marine reserves discussed in this report, can mean that fishers benefiting from the biggest fish and best catches may be those acting illegally. This can undermine the confidence and support of those who are honest. In an economic assessment of the SMMA in St Lucia, Clark (2002) estimated that if growth rates in fish catches observed during the first 5 years of management were sustained over the next 15 years (a possibility suggested by findings from Apo Island, Maypa et al. 2002), reef fishers' incomes would eventually rise above the St Lucian poverty line.

Positive social effects of establishing marine reserves include increased environmental awareness and educational opportunities amongst local communities. Marine reserves often form a focus for conservation activities which spread into more general practice. The initial consultation process often includes a strong educational element. The SMMA in St Lucia offers school programmes and takes local people out to view the reef in glass bottom boats. At Apo Island, revenue generated by the marine sanctuary was used to provide scholarships to local students (Bernando 2001). Park offices that double as information centres for local use allow people to share park resources.

Direct involvement of local communities in monitoring marine reserves can increase people's interest and enthusiasm for them. The Eastport lobster closures in Newfoundland involved local schools and other community members in collecting and analysing lobster landings to assess the effectiveness of closed areas (Rowe and Feltham 2000, see case study). Community monitoring has also been successful for invertebrate fisheries in Fiji (Tawake et al. 2001, see case study). Neis (1995) suggests that information and interpretation centres could become places of education for fishers, and somewhere for them to disseminate their traditional knowledge to communities, tourists, visiting researchers and policy-makers. Involving recreational and commercial fishers in tagging programmes to study movements of fish has also been successful, for example with recreational linefish in South Africa (Bullen et al. 2000). Whenever communities are involved in monitoring, the regular feedback of research findings is important to success (Tawake and Aalbersberg 2001, Horrill et al. 2001). Obura et al. (2002) found that community fisheries monitoring was fundamental to management in the Kiunga Marine National Reserve in Kenya. Participatory monitoring increases awareness of the need to manage resources and allows the community to contribute (Obura et al. 2002).

In some cases marine reserves have increased community harmony and reduced conflicts between user groups and managers when people have had to work together on committees and make joint management decisions. Committee-based management can also act as a social leveller whereby fishers, tourist business

operators and local politicians interact on the same level to make decisions. In Fiji, Tawake et al. (2001) reported that the success of closed areas for invertebrate fisheries had led to community pride and confidence in the strength of people to bring about change. This community became a role model for others in the region and sent advisors out to other projects.

However, there are also reports of marine reserves exacerbating existing problems. One of the most common conflicts is between fishers and recreational users, particularly divers. Where people are permitted to dive in marine reserves, fishers can sometimes feel that they are being unfairly excluded, but others remain unregulated. These kinds of resentments have been voiced in the SMMA in St Lucia (pers. obs.) and in the Florida Keys (Suman et al. 1999). Problems can also arise where itinerant fishers use areas on a sporadic or seasonal basis. Itinerant fishers are not usually consulted about marine reserves because they are not seen as part of the local community. However, they may have a significant impact on local resources. Displacing them from reserves can lead to problems elsewhere, as happened when fishers moved from the Nosy Atafana marine reserve in Madagascar to other coastal areas (Grandcourt et al. 2001). Alternatively, they may continue to fish illegally, often because they are not subject to the same social pressures as the immediate community. Grandcourt et al. (2001) suggest that the problem of itinerant fishing requires improved and co-ordinated coastal zone management across a wide area. The problem of defining a local community was illustrated in planning the Te Whanganui-a-Hei Marine Reserve in New Zealand. Here one local community was targeted for consultation, but people from other nearby towns who also used the area regularly were not included and felt that they should have been (Cocklin et al. 1998).

During initial consultations the full cross-section of the fishing community should be represented. In any community initiative there is danger that decision-making will be dominated by strong personalities representing minority points of view rather than by broad representation of all stakeholders. Even when user groups are all represented on a committee, the views presented may be limited to a much smaller sub-group, or to the individual interests of the representative. If this is allowed to happen, groups that may already have been marginalized because of social and economic differences may become further isolated and disenfranchised. Groups that are often overlooked or viewed as part of another group in which they have no voice include poor subsistence fishers and women fishers. This may arise from cultural restrictions or because such people are more involved in fishing for household use rather than commercial. Fishing for home consumption is often given less value by decision-makers. Users who speak a different language or come from a minority ethnic group may also find themselves excluded because they find it hard to access information, e.g. Spanish speakers in the Florida Keys National Marine Sanctuary (Suman et al. 1999). Fishers whose activities are already perceived as socially unacceptable or part of the management problem, for example dynamite or poison fishers, may also be excluded from management. Such omissions can lose vital opportunities to accommodate these people in management plans, and usually results in problems remaining unresolved.

In some areas where marine reserves have been established, fishers see their exclusion as evidence that they are being blamed for deteriorating resources, whereas they feel that poorly managed coastal development, pollution and other external influences are more the cause of decline. If fishers have this perception they will feel unfairly treated and be unlikely to support reserves. The role of education and liaison with fishers in such cases is obviously vital. The other impacts contributing to fisheries decline such as habitat damage and pollution will need to be acknowledged and addressed in educational approaches to the fishing and broader communities.

13.1 Alternative income generation

Marine reserves can also help generate alternative incomes for fishers. For example, one of the most effective ways of integrating local fishers' knowledge into the management of marine reserves is to employ them as park staff. This has proven effective at Nabq in Egypt where local Bedouin have become community rangers (Galal et al. 2002). In the SMMA in St Lucia all the rangers are recruited from the local fishing community and their knowledge is invaluable in enforcement.

Agriculture projects were used in management of the Nosy Atafana Marine Park in Madagascar to provide alternative and additional sources of income for displaced fishers (Grandcourt et al. 2001). As part of a community-based coastal zone management plan in Tanga, Tanzania, a number of alternative livelihood options were tried. Problems were encountered with fish aggregation devices, oyster farming and fish farming but seaweed farming was successful and now involves fishers from many communities throughout the district (Horrill et al. 2001).

Tourism focused around the reserves in St. Lucia has opened up economic opportunities for fishers to use their boats as water taxis (F.R. Gell and C.M. Roberts, unpublished data). This in turn has reduced pressure on the fishery, providing alternative employment for people who would otherwise have to fish. Improvements in fish populations in reserves can attract more tourists and improve the ability of local communities to capture tourism revenue. Rudd (2001) and Rudd and Tupper (2002), for example, show how tourists in the Turks and Caicos would be willing to pay more for their dives in places with large Nassau groupers and spiny lobsters (*Panulirus argus*). Sala et al. (2001) suggest that Nassau groupers are worth more in the water in Belize than as

meat, based on possibilities of tourism centred on the spawning aggregations of these fish. In Kenya, Malleret-King's (2000) finding of greater household food security among people living close to the Kisite marine reserve than those living further away, was largely because of their ability to generate revenue from tourists (see case study).

However, in some areas fishers have limited opportunities to benefit from direct employment in the tourist trade. In Montego Bay, Jamaica, for example, hotels did not hire fishers because they "did not match the clean cut profile of most hotels" (Bunce et al. 1999).

In areas where dive tourism is important, fishers displaced by marine reserves can be given opportunities to retrain for work in the dive industry. As part of the establishment of the Seaflower Biosphere Reserve in Colombia, 120 local people, including fishers, were given the opportunity to learn to dive and ten of them are going on to become professional divers. One other advantage of this approach was to get fishers and dive operators working together and resolving conflicts (M. Howard pers. comm.).

A number of studies have identified the importance of maintaining a diverse source of incomes for artisanal fishers in societies where this is the norm (Allison 2001, Fiske 1987, Howard pers. comm.). Obtaining income from a combination of fishing, farming and other opportunistic sources of income such as trading or casual work is common in many developing countries. People used to this approach can often not adjust to regular full time work in the tourist sector. Diverse sources of income also buffer people from one particular activity failing.

Fishers displaced by marine reserves may be able to develop other fishing opportunities. For example, in St. Lucia an offshore fish aggregation device allowed some reef fishers to switch to catching open water species. In this and other areas, loans have also helped people purchase bigger boats and engines, which again encourages them to fish for offshore pelagic species.

Some of the most effective reserve areas that we present in the case studies accompanying this review have been small-scale local management initiatives, such as the Local Marine Management Area approach in Fiji (Tawake et al. 2001, Tawake and Aalbersberg 2001), the Eastport lobster management programme in Newfoundland (Rowe & Feltham 2000) and the SMMA in St Lucia (Roberts et al. 2001a, Gell et al. in prep. a). Other successful cases featured have been more top-down and part of larger scale management initiatives, including Georges Bank (Murawski et al. 2000), South African marine reserves (Atwood et al. 1997) and the Florida Keys National Marine Sanctuary (Bohnsack 2002, Suman et al. 1999). The effectiveness of each approach depends very much on the situation under consideration. Through work with a marine management area project in Indonesia, J. Alder (pers. comm.) found that small-scale community-based management can be successful if the threats to resources are small and local. However, larger scale management is necessary when there are significant external threats such as fishers from elsewhere trawling close to shore.

Methods of enforcement for marine reserves vary greatly depending on the type of fishery and the area involved. Effective compliance is obviously essential for success, so it is important to identify the most effective approaches and what works best for local communities. In applying penalties for illegal fishing in subsistence fisheries, managers have to consider the reliance of people on fishing for their survival. Under these circumstances, the destruction of fishing gear or use of unrealistic fines would lead to increased poverty and hardship. On the other hand penalties have to be advertised and used when appropriate to support compliance. If people are seen to be getting away with illegal fishing, it can undermine management completely. In societies with strong social pressure the shame associated with poaching, which is in effect cheating the whole community, is a powerful deterrent.

In an ideal reserve, very little enforcement would be needed, as a supportive community would comply with rules and police each other's activities. Alder (1996) identified education as the key management tool for marine protected areas, and recognized the importance of combining enforcement and education. In a study on the Great Barrier Reef in Australia she found that per person, education was cheaper to implement than enforcement. Effective education can foster self-enforcement, as is the case in San Salvador Marine Park in the Philippines where local residents who understand and support management assist in enforcing regulations and reporting illegal fishing (Buhat 1994).

Compensation for fishers in the early years of reserve management is a controversial issue but one that often needs considering. Compensation may be necessary where fishers have lost a significant proportion of their former fishing grounds and where options for fishing in other areas are limited. These might include geographic constraints or prohibitive fuel costs. Compensation may also be necessary when fishers have limited opportunities to develop alternative incomes. However, compensation can only be a short-term fix, and theoretical models of reserve effects suggest that temporary measures should be all that is needed. Such models predict an initial drop in catch per unit effort immediately after reserve establishment, a pattern supported by our St. Lucian data (Gell et al. in prep. a). Compensation can help ease a community through the difficult initial steps towards eventual fisheries benefits. Without it, this stage may never be reached.

Campellao and Georgiadis (1995) discuss the issue of compensation and suggest that group compensation is generally preferable to individual compensation because of the difficulty of identifying who is

entitled to what. One of the barriers to direct compensation for fishers is a general lack of economic data on costs and benefits of fishing.

Beaumount (1997) points out that in cases where it might seem preferable to award compensation in the form of a long-term, community development project, this could be undermined by the very real needs of individual households who are suffering as a result of a marine reserve. Difficult decisions may have to be made in situations where people are subsistence fishers living at, or below the poverty line. This was the case among reef fishers in St Lucia, where the oldest (most of them above 50) were given compensation payments of US$150 per month for 12 months (Roberts and Hawkins 2000). This financial help greatly improved their compliance and support for marine reserves, paving the way for eventual catch enhancements.

14. Conclusions

The evidence that marine reserves can benefit fisheries is growing stronger. There is abundant evidence that reserves support increases in populations of exploited species. They have been effective for a wide diversity of species across almost the whole range of taxa targeted by fishers, including fish, crustaceans, mollusks, echinoderms and ascidians. Reserves can be designed to benefit animals with a broad spectrum of life-history, behavioural and movement characteristics. Changes in size and age structure of protected species foster rapid rates of increase in reproductive output, with order of magnitude increases in egg production regularly reported from reserves. Effects of protection develop rapidly with increases of 2-3 times in biomass often achieved within a few years. However, benefits continue to accrue over long timescales as habitats recover and populations develop extended age-structures. Reserves have worked well in many different habitats and geographic regions of the world.

There are now many studies showing local enhancement of fish catches close to reserve boundaries. There are also a growing number of cases in which overall catches have increased to levels greater than when there were no reserves and fishing grounds were larger. Fishery effects tend to develop quickly and have frequently been detected within five years of reserve establishment. Those effects are greatest (and easiest to detect) in exploited areas close to reserves, but in most cases larval export probably extends benefits beyond the regions studied. Small reserves work well, but regional effects on landings require reserves to be scaled up and established in networks. Some of the most convincing success stories are from places where reserves have protected between 10 and 35% of fishing grounds. There are also cases where reserves have sustained fisheries at intensities that would otherwise have caused their collapse. Recreational, artisanal, small-scale and industrial fisheries have all benefited from reserves or fishery closures in places where management ranges from self-policing by committed fishers through to satellite monitoring.

The experiences we have described show that success of marine reserves is not contingent on habitat type, geographical location, the kind of fishery involved, or the technological sophistication of management. We now have strong evidence that with the support of local communities, marine reserves offer a highly effective management tool. Of course, every region will pose new challenges, and every experience will offer new insights and lessons. However, fear that reserves will fail to have any biological or societal benefits is not something that should keep fishers or would-be reserve managers awake at night.

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Part 2: Case Studies

1. The Nabq Managed Resource Protected Area, South Sinai, Egyptian Red Sea

The Nabq Managed Resource Protected Area is a multiple-use management area that covers 47km of coast on the east of the Sinai Peninsula in the Egyptian Red Sea. It has a total marine area of 122km^2 and also includes 465km^2 of adjacent terrestrial habitats. It was established in 1992 by the Egyptian Environmental Affairs Agency and forms part of the larger network of five parks and protected areas of the Southern Sinai Protectorates: a total area of 8,085km^2 of land and 1,651km^2 of sea (Pearson and Shehata 1998). The area is characterised by well-developed fringing reefs running parallel to the coast and lagoons which are often dominated by seagrass beds. There are also four areas of mangrove (the most northerly in the Indian Ocean region). The local fishery is conducted by Bedouin who are semi-resident at two small coastal camps within the Nabq Protected Area. It is a multi-gear, multi-species fishery, employing mainly trammel nets and gill nets from the shore, with some handline fishing. The area is protected from large-scale development, and destructive and unsustainable uses of resources are not permitted. For example, destructive fishing techniques such as dynamite fishing and trawling are banned throughout the protected area. This is a precautionary measure as these techniques have never been used locally. There is also a spearfishing ban and minimum mesh size regulations on nets.

Discussions with local fishers and community leaders built up local support for the management area, and a key development in community participation was the employment of local Bedouin as 'Community Rangers', involved in day-to-day management. Community support and involvement was well established by 1995 when a network of fully protected areas, in which all extractive uses are forbidden, was incorporated into the management plan. These areas were selected in close consultation with local fishers, and the process was accompanied by community education on ecology, conservation and management issues. Support for conservation was strong and built upon the Bedouin's existing traditional conservation measures, for example the return of bycatch to the sea and a ban on cutting timber for fuel and building. Five no-take zones were established in a network with fishing grounds interspersed between them. The total area designated as no-take was approximately 17km of coastline, out of a total length of 47km. The smallest area covers less than 1.5km of coastline, and the largest over 3km.

Using underwater visual census, fish abundance was estimated in 1995 when no-take reserves were established, then two years later in 1997 and again in 2000 (Galal et al. 2002). Abundance of the three main target fish families was significantly greater overall after 5 years of protection (Figure 1). Of the species assessed individually, five showed a clear increase in mean lengths over the 5 years of protection. These were the emperors (Lethrinidae) *Lethrinus mahsena* and *Monotaxis grandoculis*, and the groupers (Serranidae) *Cephalopholis miniata* (coral hind), *Varioli louti* (lyretail grouper) and *Epinephelus tauvina* (greasy grouper). Abundance of the following species increased with time protected: *Lethrinus obsoletus* (orange-stripe emperor), *Lutjanus bohar* (red snapper), *Cephalopholis argus* (peacock grouper), and *Variola louti*. Abundance of *Cephalopholis hemistiktos* (halfspotted hind) and *Epinephelus fasciatus* (blacktip grouper) increased from 1995 to 2000, but dipped in 1997.

Mean catch per unit effort (CPUE) from all fished sites combined increased between 1995 and 1997, but was not statistically significant. In 2000, mean CPUE had increased further, and this time was significantly greater than CPUE at the onset of protection in 1995 (see Figure 2). Overall the increase was from 0.79 kg net^{-1} h^{-1} to 1.31 kg net^{-1} h^{-1}, a rise of 66%. When the CPUE for each of the nine fishing sites were assessed separately, seven had significantly higher CPUE in 2000 than in 1995. The biggest increases in fish family biomass seen in the reserves were at the two sites which were most heavily fished prior to the closure of the reserves. The increases in the length of fish were also most dramatic at sites which had been heavily exploited by hook and line fishing, rather than net fishing, presumably because line fishing selects for the larger individuals.

There was no change in the number of fishers active in the area over the time period so the increase in CPUE also suggests an increase in the total catch, despite the loss of fishing grounds to reserves. Overall the Nabq area was not classed as overfished before the implementation of the marine reserves. The study indicates that reserves can have benefits even in areas where a relatively small fraction of high value species are targeted, contrasting with intensively fished places like Apo Island in the Philippines or the Soufrière Marine Management Area in St. Lucia (see other Case Studies).

The management history of the Nabq Marine Protected Area may also have contributed to improvements in the fishery, and the increased abundance of key commercial families. The ban on damaging fishing methods that has been in place in the protected area since 1992, and the use of traditional management techniques and the increased awareness of environmental issues locally have all been important in the successful management of the
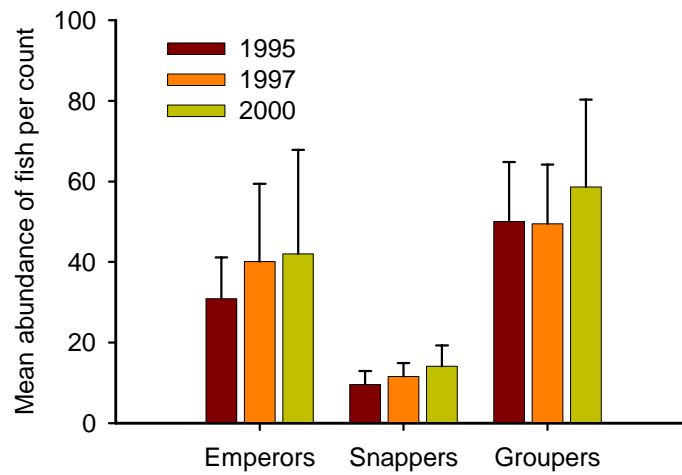
Figure 1: Mean abundance of the three main commercially important fish families in marine reserves at Nabq. Data are pooled from all ten stations in marine reserves. Abundance values are per 8000m^2 transect. Error bars show +/- 1 SD. Mean abundance has increased significantly for all three families (Wilcoxon signed-ranks test P<0.05). Reproduced from data in Galal et al. 2002.

fishery. Compliance with reserve protection has not been perfect, and some of this is due to inadequate marking of the reserve boundaries. Most illegal fishing is done by boys and young men (R.F.G. Ormond, pers. comm.).

Much time was invested in community discussions and consultations in the year prior to the establishment of the reserves and this was regarded as vital to its success. The community rangers were particularly important. Not only did they play a role in enforcement but they were also important in monitoring effectiveness of management through collection of CPUE data. There were two keys to the successful design of the reserves at Nabq. Firstly the locations of the reserves were decided in consultation with the fishers, taking into account traditional use, access to sites and other social factors. Secondly, the network design interspersing five small reserves with fishing grounds allowed higher potential for spillover across reserve boundaries than one large protected area. This arrangement also allowed a broader variety of habitats to be protected by the reserves, encompassing examples of the reef, seagrass, mangrove and sand lagoon habitats found in the area.
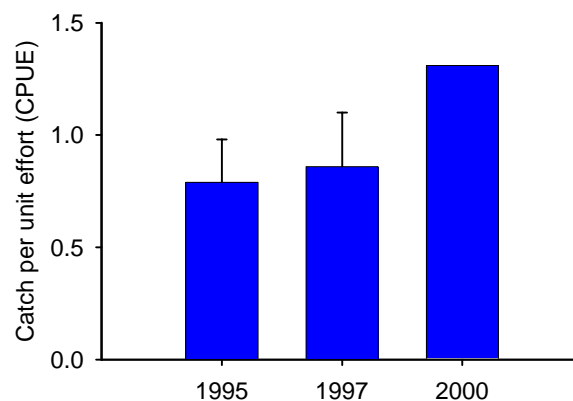
Figure 2: Mean catch per unit effort for fishing grounds adjacent to marine reserves at Nabq. CPUE is in kg per net per hour, and the error bars show +/- 1 SE. Produced from data presented in Galal et al. 2002.

Further work is now going on to investigate the effects of no-take reserves on non-target reef fish, in addition to the regular monitoring of fishing effort and catch. There is also an ongoing project looking at the effects of the closed areas on the local reef-flat invertebrate fisheries. Preliminary results indicate higher abundances of invertebrates such as giant clams (*Tridacna* spp.) and other molluscs within the no-take areas compared to other areas (R.F.G. Ormond and J. Ashworth pers. comm.).

Key points

- Fully protected areas were implemented within an established management area and were designed and managed in close collaboration with the community.
- Community rangers were highly successful in achieving compliance with regulations.
- Fish from three key commercial families – groupers, emperors and snappers – increased in abundance in the reserve after 5 years of protection and catch per unit effort in the adjacent fishery increased by two thirds.
- The establishment of the network of no-take marine reserves has played a key role in maintaining the sustainability of the fishery.
- Other management tools complemented the fully protected reserves, including traditional measures such as return of bycatch, and a ban on spearfishing.
- The Nabq area shows that fully protected reserves can offer benefits in places that are not intensively fished.

Thanks to Nasser Galal, Jennifer Ashworth and Rupert Ormond.

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2. Contrasting experiences from the Philippines: Apo and Sumilon Islands

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The Philippines has a long history of the management of coral reefs and other coastal habitats using marine protected areas, and now has a National Integrated Protected Area System, established in 1992. In a 1999 survey, Pajaro et al. (1999) identified 439 MPAs in the Philippines, with 44 being fully enforced. The 44 enforced areas cover an estimated 26,500 hectares (265km^2) and mainly protect coral reefs, encompassing just under 1% of Philippine coral reef habitats. Nationwide fishery catch per unit effort in the Philippines has decreased dramatically over recent years (White et al. 2002), but increases in catch per unit effort and fish yields have been reported from a numbers of fisheries adjacent to marine reserves (see case studies below, and San Salvador Island, White et al. (2002), Christie and White (1994) and Garces and Dones (1998), in White et al. (2002)).

Apo

Apo Island in the central Philippines has an area of 0.74km^2 (Maypa et al. 2002) and in 2001 had a population of 700 (Maypa unpubl. data). Management began at Apo in 1976 when staff from Silliman University, on the nearby island of Negros, introduced a marine conservation and education programme. The idea to have a no-take area evolved from this programme and in 1982 the community began protecting an area that is now known as the Apo Reserve or Marine Sanctuary. The area is a 400m long section of reef along the southeast coast of the island, protecting about 10% of the total reef area (Figure 1). In 1985, as part of the Silliman University Marine Conservation and Development Programme, a marine reserve plan was endorsed by the community which included the 400m no-take section and also designated all the island's reef to 500m from shore as protected. Management responsibility was given to the Marine Management Committee, a group of local residents. The main objectives were to prevent non-residents fishing the island's reefs, to prevent the use of destructive fishing techniques, to protect habitats and breeding fish, to increase fish numbers in the no-take area and therefore local fish catches by export of fish, and to encourage tourism. The area became formally protected through legislation in 1986 (Russ and Alcala 1999).

Figure 1: Map and photograph showing the location of the Apo Reserve. The map shows the location of fish visual census transects (dashed areas) studied by Russ and Alcala. Photo by R. Raymundo and layout by J.L.P. Maypa.

White and Calumpong (1992) carried out surveys of fishers' opinions about the reserve in 1986 and in 1992. All 21 fishers interviewed (representing 20% of Apo fishers) supported management of the area and the no-take reserve. By 1992 all the fishers interviewed had perceived an increase in their catch since implementation of management. Some fishers claimed their catches had doubled over the 10 years of protection. Most fishers interviewed also said that the management programme had led to increases in tourism, although less than half (10/21) felt they were benefiting from tourism. The same number said that tourism was having negative effects, particularly dive boats arriving from other islands and damaging the reefs with anchors. In 1992 there was also evidence for good compliance with fishing regulations and the reserve (White and Calumpong 1992), and throughout its 20 years of management Apo has been known for its strict enforcement and good compliance (Maypa et al. 2002).

In a recent paper, Maypa et al. (2002) bring together a total of six assessments of fish yield and CPUE studies conducted at Apo between 1980 and 2001. They used three assessments of their own from 1997/8, 2000 and 2001 and recalculated data collected in other studies in 1980/1, 1985/6, and in 1986 (Alcala and Luchavez 1981, White and Savina 1987, Bellwood 1988) to allow the values from each study to be fairly compared.

Over the past 20 years there have always been 100 or more fishers on Apo (111 in 1987 (White and Savina 1987), 119 in 2001 (Maypa et al. 2002)). Since the 1980s more than 90% of the total fish yield has been taken by hook and line, although gill-nets, spears and bamboo fish traps are also used. Reef-associated yield was maintained at between 15-30 t km^{-2} year^{-1}, but non-reef catches (which have not benefited from reserve protection) declined from 6.21 t km^{-2} year^{-1}, to 1-2 t km^{-2} year^{-1}. Line fishing catch per unit effort increased 10-fold from 0.13-0.17 kg man^{-1} hour^{-1} in 1980/1 to 1-2 kg man^{-1} hour^{-1} in 1997-2001.

Five fish families dominate Apo catches, the most important being Carangidae (jacks) and Acanthuridae (surgeons). The importance of these two families increased over the study period from a total of 40% of the catch in 1980/1 to more than 70% in 2000/1, and reef-associated fish species in general also grew more important. Jacks accounted for 26% of the catch in 1980/1 and for 47% in 2000/1. This suggests that the Apo fishers have become more reliant on reef species over the years and they have been able to catch more (Maypa et al. 2002).
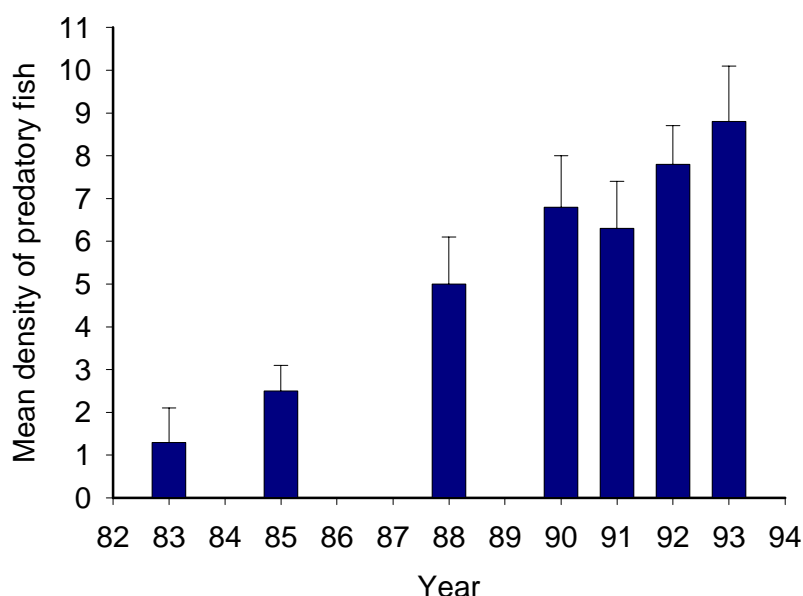
Figure 2: Mean density of predatory fish (groupers, snappers and emperors) in number per 1000m^2 (+/- 1SE) in the Apo reserve from 1983 to 1993, estimated by visual census. Redrawn from Russ and Alcala 1999.

A striking feature of this long-term study is that the exceptionally high fish yields of Apo Island have been maintained for twenty years. The evident sustainability of the fishery is attributable to the combination of a well-respected reserve and the prohibition of damaging fishing methods. Apo's success is in stark contrast to the generally declining catches in the Philippines, where total catches have failed to increase despite increasing effort in terms of boat numbers and sizes and in numbers of fishers (BFAR 1997, Courtney et al. 1999 in White et al. 2002). Catch per unit effort in the Philippine small pelagic fishery in 2000 was less than a tenth of what it was in 1948 (BFAR 1997 in White et al. 2002).

At Apo, total catches are stable while catch rates have increased, suggesting that fishing effort has declined over the study period. We need to take this into account when interpreting the changes observed. Apart from the lack of reserve protection, one reason the non-reef catch declined was that use of drift gill-nets decreased, and completely stopped in 1999/2000. Bamboo fish traps were commonly used from 1980-6 but were not used at all in 2000 because of problems with tourists damaging them. Their use resumed in 2001.

Russ and Alcala (1996) assessed changes in predatory fish density between 1983 and 1993. Inside the reserve there was a steady increase in density, particularly of snappers and emperors (Figure 2). An update on fish densities is forthcoming, and the trend of increase apparently is continuing (G.R. Russ pers. comm.).

Despite setting aside 10% of the reefs where they previously fished, fishers in Apo appear to be benefiting from the no-take area, with line fishing CPUE increasing by an order of magnitude. There have been other benefits too. Two tourist resorts were developed on Apo in the 1990s and many visitors are attracted by the marine reserve. Apo has other dive sites, but the reserve is internationally known and is where the tourists want to dive (Bernando 2001). Alcala (1998) estimated that through reef-associated tourism, the local economy was benefiting by US$500 per hectare of reef per year (taking into account all the reef area of Apo, not just the reserve).

Bernando (2001) gives further insight into the effects of the reserve in his study of environmental impacts and distribution of benefits. He assessed socio-economic indicators such as household income, fishing income, skills development, reserve income, environmental awareness, management capacity and capability to enhance or generate local policies. All had improved since 1984. He estimated benefits from the reserve to be in the region of US$200,000 in 2000, but noted that much was going to dive shops and resorts rather than directly to the community. However, the local community benefit from tourism not only by being employed in the resorts, but also through independent enterprises such as the sale of t-shirts and other souvenirs. His interviews showed that fishers perceived their catches increasing in response to protection up until 1992, then declining, particularly in the past five years (1995-2000) whereas Maypa et al. (2002) show that CPUE continued to increase until 1997/8 but had decreased in 2000/1. Perceived decreases in catch corresponded to the establishment in 1991 of the first tourist resort on the west of the island (a second resort was built near the main village in 1996). The present management authority of the reserve, the Apo Island Protected Landscape/Seascape (APLS) Protected Area Management Board, collects visitor entrance fees and levies fees for diving, snorkeling, and mooring and anchoring in the area. From the income generated from fees 75% goes back to the community for the management of the reserve and for community projects, whilst 25% goes to the Protected Area Management Board and the Department of Environment and Natural Resources.

Important factors in the success of Apo include training opportunities for fishers, women and young people of the Apo community and the reliable and continued technical support of the Silliman University since the reserve was established (pers. obs. A. Maypa). The establishment and maintenance of a marine education and community centre has also provided a focus for the management and maintained enthusiasm in the area, and provided an "immediate and tangible benefit" for community involvement (Russ and Alcala 1999).

In Russ and Alcala's (1999) assessment of the management success of Apo, they conclude that all of the stated objectives for the area have been fulfilled. Fishing is almost entirely confined to local people, destructive fishing techniques are not used, there has been a build up in fish biomass and there is strong evidence for the export of adult fish to adjacent fished areas, and a successful local tourism industry has developed.

Sumilon

The early history of the Sumilon Reserve, also in the Central Philippines, about 44km away from Apo, is similar to that of Apo. However, important differences are that Apo has a resident community whereas Sumilon is uninhabited and is fished by people from several communities. In 1973 biologists and social scientists from Silliman University began a marine conservation and education programme with the southern Cebu communities who fished Sumilon. The idea for a marine reserve evolved out of this programme. Scientists chose Sumilon because it had relatively good coral cover and fish, and one of the objectives of the reserve was to increase fish yields. In 1974 the Sumilon Reserve was formally established through a local government ordinance, with an agreement between Silliman University and Oslob Municipal Council (that has jurisdiction over Sumilon). The agreement gave Silliman University authority "to establish a marine park around the island for marine biological studies and research", and a no-take area was established. Its objectives were very similar to those at Apo, including protection of fish habitat, fish biomass build-up in the reserve and export of fish and their offspring to enhance catches. It was also designed to encourage tourism in the area.

The reserve was enforced by a caretaker who was a local fisher, and he also monitored fish yields in adjacent areas. A study of fishers' perceptions in 1976 by social scientists from Silliman University revealed that many did not really understand the purpose of the reserve (Cadeliña 1976). Despite this, in the late 1970s and early 1980s many fishers reported increased catches (Alcala 1981, 1988, White 1988).

In 1980 new mayors were elected in Oslob, and in Santander, another major community fishing around Sumilon. Part of their election campaigns was to open the Sumilon Reserve and shortly afterwards there were a number of serious cases of illegal drive net fishing in the reserve. In response, Silliman University appealed to the national government and Sumilon Reserve was declared a national fish sanctuary. This moved responsibility for the reserve from the local community to the national Bureau of Fisheries and Aquatic Resources (BFAR) and caused widespread local resentment. Intermittent illegal fishing continued from 1980 to 1984. In 1984 regular fishing began in the area and the caretaker was removed for his own safety. Fishing included damaging methods such as drive net fishing and dynamite fishing and persisted until the end of 1985. The Mayors then decided that they should comply with the national BFAR order (probably because there were plans for tourism development on Sumilon) and fishing stopped in the reserve in 1987 (Russ and Alcala 1999).
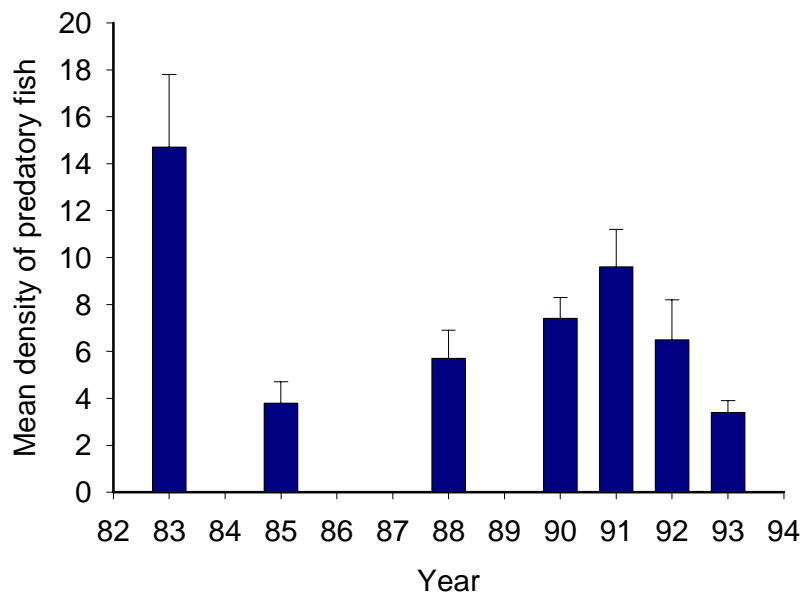
Figure 3: Mean density of predatory fish (groupers, snappers and emperors) in number per 1000m^2 (+/- 1SE) in the Sumilon reserve from 1983 to 1993, estimated by visual census. Redrawn from Russ and Alcala (1999).

In 1988 the Oslob Municipal Council banned all fishing on the whole reef of Sumilon. The area was protected for 4 years until 1992 when the resort on the island was completed. Once the resort was operational fishing was permitted in non-reserve reef areas, but with no enforcement or surveillance the whole area was soon fished once again. In 1994 a renewed attempt to protect the area was made and by 1997 a caretaker was once again in place to enforce regulations. However, this new caretaker tolerated some illegal trap fishing and allowed hook and line fishing in the reserve. By 1997 the tourist resort had become disused – tourism had not brought the income to the area that had been anticipated. The use of hook and line and trap fishing in the reserve has persisted to 2001 (Alcala, submitted). Figure 3 shows how densities of large predatory reef fish in the Sumilon reserve have fluctuated in response to the changing management regime (from Russ and Alcala 1999). Trap yields have also varied over the years (Alcala, submitted).

Sumilon is an example of how things can go wrong with marine reserves. Politics and conflicts between local and national management approaches led to failure. Community involvement was not gained fully at the outset and there was misunderstanding of the motives of the University. Expectations of the fishery benefits of the marine reserves amongst fishers may also have been unrealistically high. However, Sumilon provides a very valuable demonstration, replicated over time, of the rapidity of stock and fishery response to reserve protection, and of the speed with which gains can be eliminated when fishing is renewed.

Key points

- Very high reef fishery yields have been maintained at Apo Island over a period of more than 20 years, attributable to a well-respected reserve which covers 10% of former fishing grounds, and a ban on destructive fishing methods.
- Since reserve creation, Apo line fishing CPUE increased 10-fold.
- Apo had a resident community so the sense of ownership and responsibility for resources was clearer than at Sumilon which was uninhabited.
- Apo and Sumilon show that small reserves can benefit local fish catches.
- The Sumilon example shows that increased catches and fish densities can be achieved by protection, but that these gains can be lost quickly if protection lapses.
- Sumilon emphasises the importance of community involvement, and shows that support of local politicians can be as critical as grassroots support.

Thanks to Angel Alcala and Garry Russ.

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3. The effects of New Zealand marine reserves on exploited species and fisheries

New Zealand was one of the pioneers of marine reserves in temperate waters, having had functioning fully protected marine reserves since 1977. There are now 15 marine reserves in New Zealand protecting approximately 4% of its territorial waters (Department of Conservation, www.doc.govt.nz/Conservation/Marine-and-Coastal/Marine-Reserves/). All of them are fully protected from fishing, and all have been so from the outset, with the exception of the Poor Knights Reserve, which was initially opened to limited recreational fishing, but closed again after a ministerial review (Cocklin et al. 1998). New Zealand's marine reserves protect a variety of habitats (including the subtropical Kermandec Islands). In the longest established and most intensively studied marine reserves and adjacent areas in the north east of New Zealand, the characteristic habitats are temperate reefs, with kelp, patch reefs and soft sediment.

Leigh Marine Reserve

The Leigh Marine Reserve encompasses 5km of coast, extending 800m seaward on the north east coast of New Zealand. The reserve was gazetted in 1975 after a 10-year process of application and consultation. The initial motivation for its establishment was the concern of scientists from the Leigh Marine Laboratory of the University of Auckland over the level of exploitation of the shores and coastal waters, particularly from spearfishing. Its main aims were conservation of the marine environment and scientific research. The reserve became actively managed in 1977 and is enforced by marine rangers and by community enforcement via the Department of Conservation. Although not initially established for fisheries management, the fishing community have come to support the reserve, as have members of the wider community who have seen benefits through the increases in visitor numbers and associated revenue (Walls 1998).

In a survey after 10 years of management (Crouch and Hackman 1986, in Ballantine 1991) 78% of commercial fishers said they were in favour of more reserves, 78% said they would actively prevent poaching in the Leigh reserve and 40% said that their catches were higher because of the existence of the reserve.

A second survey was conducted at Leigh in 1992 (after 15 years of active reserve management), focusing on three user groups – visitors, local residents and local businesses and this demonstrated almost total support. Significantly, the survey confirmed the reserve had the support of commercial and recreational fishers, and that many fishers believed that fishing in the adjacent fishing grounds had improved. The role of fishers in enforcement of the reserve regulations was also found to be important in the day-to-day management (Cocklin et al. 1998).

Public participation and support is not currently a statutory requirement in the designation of marine reserves in New Zealand, but is now included in the application process (Department of Conservation 1994, in Cocklin et al. 1998). In the case of Leigh, Cocklin et al. (1998) identify a problem in defining the "local community" in that all consultation focussed on one community. Another nearby community felt they too should have been involved. The importance of including visitors in consultation is also raised here.

The history of Leigh Marine Reserve shows that even when there has been long-term consultation, full agreement might not be reached at the outset. However, one encouraging characteristic of Leigh is that support has increased over time and now few people have objections. In a socio-economic study many local people believed that the community had benefited economically from the presence of the reserve, mainly through visitors buying food (Cocklin and Flood 1992, in Walls 1998). The indigenous people were not specifically included as a group in the initial consultation, although individual Maori people were involved. New reserves in New Zealand now specifically consult indigenous people under the Treaty of Waitangi (Walls 1998).

Effects on lobsters and the lobster fishery

In the Leigh Marine Reserve, Kelly (1999) found that catch rates of lobsters showed strong seasonal variability. However, catch rates close to the reserve boundary were high compared with areas further away. Local lobster and fin fishers also choose to fish close to the boundary implying that the public perceive that the reserve has increased the abundance of fishery species (Kelly 1999).

Kelly et al. (2000) assessed recovery of the spiny lobster *Jasus edwardsii* in four marine reserves in north eastern New Zealand and compared this with similar non-reserve sites. They included Leigh Marine Reserve (protected for 21 years), Tawharanui Marine Park (protected for 14 years), Cathedral Cove Marine Reserve and Tuhua Marine Reserve (both protected for 3 years). They found higher lobster biomass inside the marine reserves than outside and they were also able to look at the extent of lobster recovery in relation to time since protection.

Lobster densities inside reserves increased by nearly 4% per year in shallow sites (less than 10m) and by 9.5% in sites deeper than 10m. Mean carapace length of lobsters increased by 1.14mm per year of protection, and lobster biomass was estimated to have expanded by 5.4% per year of protection in shallow sites and by 10.9% per year in deep sites. Estimated egg production increased by 4.8% (shallow) and 9.1% (deep) per year of protection.

In 1985/6 lobster fishers began setting their pots around the boundary of the Leigh reserve. Fishers reported very large catches with large male lobsters filling pots (Kelly et al. 1997, in Walls 1998). More recently, Kelly et al. (2002) looked at the value of the spillover fishery around the Leigh Marine Reserve. They compared catch per unit effort at the reserve boundary with a fishing site 0.3-2km from the reserve and another site 22-30km from the reserve. They found no significant difference in CPUE (in kg per trap haul) among the sites. However, catches from the reserve boundary could only be made in the deeper offshore habitat as fishers could not use the inshore reefs favoured by lobsters at certain times. Catches from the reserve boundary contained fewer but larger lobsters, and were more variable than those from the other two sites. However, the amount of money made per trap haul was similar at each site because the occurrence of empty pots was offset by pots containing large numbers of lobsters. For instance, in 1995 nearly 21.6% of revenue earned by the study fishers came from just 4.4% of trap hauls. High variability in catches is not something that is usually predicted for reserves and in this case is a consequence of the aggregation behaviour of lobsters near the reserve. Because of the seasonal nature of offshore aggregation, high catch rates were only possible during 7-8 months of the year and outside these months catches were likely to be low.

Jasus edwardsii supports one of New Zealand's most valuable inshore fisheries and is managed though a quota scheme that is perceived by fishers to work fairly well. The lobster fishing industry therefore opposes marine reserves, arguing that the quota system is a conservation tool and that marine reserves are not an effective management tool (S. Kelly pers. comm.). No detailed information was available on how the fishing community in general responded to the presence of reserves. However, the study fishers from whom Kelly et al. (2002) collected the CPUE data, responded to the loss of inshore reef sites and reduction of fishing area at the marine reserve by increasing the density of traps set around the reserve boundary. Evidence from New Zealand's reserves suggest that while they are presently small and acting principally as conservation tools, they could play a useful role in supporting other lobster fishery management measures.

Effects on commercial and recreational fish

Cole et al. (1990) studied densities of a variety of fish species in the Leigh Marine Reserve using underwater visual census in 1982 and found that only one species, the red moki (*Cheilodactylus spectabilis*) increased in abundance over the initial 6 years of management. Abundance of five other species, the snapper (*Pagrus auratus*), goatfish (*Upeneichthys lineatus*), spotty (*Notolabrus celidotus*), blue cod (*Parapercis colias)* and leatherjacket (*Parika scaber*) did not change significantly in these initial years. A subsequent survey in 1988 showed increasing abundance of snapper, blue cod, red moki and rock lobsters, but no trend in the abundance of sea urchin (*Evechinus chloroticus*).

Babcock et al. (1999) studied the most common demersal predatory fish, the snapper (*Pagrus auratus*), in the Leigh Marine Reserve and Tawharanui Marine Park and found that adults were 5.8 and 8.7 times more abundant inside the reserves than in adjacent fished areas. Individuals were also significantly larger with mean lengths of 316mm inside protected areas compared to 186mm in fished areas.

Babcock et al. (1999) found significant differences in abundance of non-target species such as sea urchins (*Evechinus chloroticus*) which declined to less than a third of their former abundance in one of the marine reserves over 20 years of protection. They also found that kelp beds were more extensive in one of the reserves. This suggests fishing pressure has changed not only the mean size and abundance of target species, but also the wider ecosystem. Snappers and lobster prey on urchins which in turn graze on kelp. In a study using tethering experiments, Shears and Babcock (2002) found that predation of sea urchins was 7 times higher inside the Leigh Marine Reserve and the Tawharanui Marine Park than in unprotected areas. Growing snapper and lobster populations in reserves have helped reduce urchin densities and facilitated an increase in algal cover. Urchin barrens covered 40% of available reef in unprotected areas but only 14% in reserves. Babcock et al. (1999) estimate that primary productivity from macroalgae like kelp has increased by 58% from what is was before the Leigh Marine Reserve was established. Benthic primary productivity was also found to be much lower outside the reserves than before intensive fishing began. Overall this study reveals some of the complex interactions that influence recovery of protected ecosystems from previously high levels of fishing.

Willis et al. (in review) studied density and size of snapper inside and outside three marine reserves in northern New Zealand: Leigh Marine Reserve, Hahai Marine Reserve and Tawharanui Marine Park. Snapper is the most

important species for recreational fishing in upper North Island and one of the most important commercial fishery species (see Annala and Sullivan 1996, in Millar and Willis 1999). The abundance of snapper larger than the minimum legal size was 14 times greater in protected compared to fished areas, and egg production an estimated 18 times higher. In the Leigh Marine Reserve, legal-sized snapper were larger than legal-sized snapper in fished areas, but size differences were not significant. Snapper abundance, like that of lobsters, was highly seasonal with higher densities in autumn than in spring.

Willis et al. (2001) explored movement of snapper in the Leigh Marine Reserve by tagging fish. They found that some show site fidelity to areas only a few metres wide and can occupy the same area for a number of years in the absence of fishing. Some snapper may move long distances but in the light of the large increases in their abundance inside the reserve it seems likely that some are permanently resident in the reserve (Willis et al. 2001). It is possible that some snappers can be mobile or site-attached and that reserves select for the site-attached snapper.

Long Island-Kokomohua Reserve, Marlborough Sound

The Long Island-Kokomohua Marine Reserve was established in 1993 and a monitoring programme initiated at the outset. This looked at size, distribution and behaviour of blue cod (*Parapercis colias*), the most common edible reef fish, and a popular target for recreational fishers. Sampling was conducted with baited hooks inside and outside reserves from 1993 to 2000, and underwater counts from 1992 to 2001.

Cole et al. (2000) studied dispersal of blue cod around the reserve. They found similar densities of fish inside and outside the reserve and that the mean size of fish inside the reserve was 4cm (19%) larger than in the fished sites (21cm in fished sites vs. 25cm in the reserves). They tagged a sample of 360 fish and 73-75% of resightings of tagged fish were made within 100m of the tagging sites. Fewer resightings were made in the reserve than at fished sites, even though individuals were not removed from the reserve sites by fishing, suggesting that some blue cod in the reserve move longer distances than those in fished sites. The larger individuals in the reserve and the limited dispersal of most of the fish indicates that the reserve is increasing survivorship. Cole et al. (2000) suggest that increased sizes within the reserve and sufficient movement via spillover is making these fish available to fishers in the adjacent fishing grounds, and therefore conferring fishery benefits.
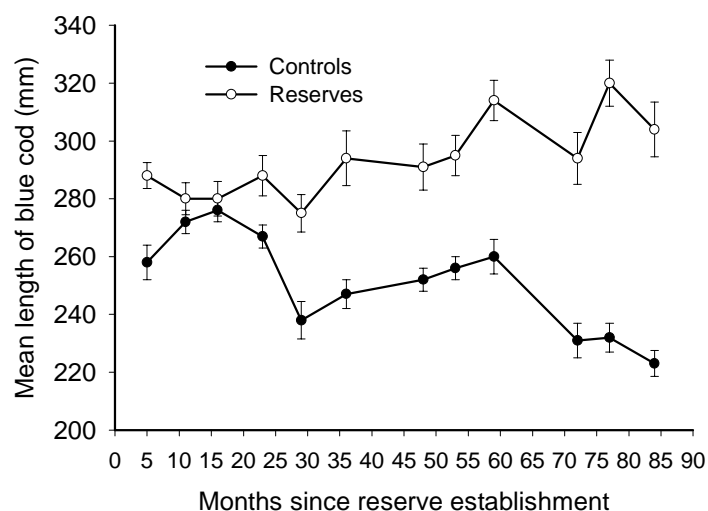
Figure 1: Mean length of blue cod from reserves and control sites from September 1993 to April 2000. Error bars show 95% confidence intervals. Redrawn with permission from Davidson 2001.

Davidson (2001) used experimental fishing with baited hooks to monitor reserve effects in the Long Island-Kokomohua Marine Reserve from September 1993 to April 2000. He found that mean length of blue cod increased from 27.6cm in September 1995 to 31.8cm in September 1999 (a 15% increase), but then decreased to about 30.5cm by April 2000. Outside the marine reserve mean length decreased from around 25.5cm to around 22cm in April 2000, probably in response to the reduction of the minimum legal size at which blue cod could be caught from 33cm to 28cm. The average fish in the reserve was approximately 39% longer than in fishing grounds after 7 years of protection (Figure 1).

Inside the reserve CPUE (blue cod per minute per rod) increased by 100% after just 11 months of protection, and continued to increase for four years after the reserve was established from approximately 0.28 blue cod per minute per rod to 1.2 blue cod per minute per rod, a four-fold increase. By contrast, CPUE outside the reserve remained approximately constant throughout the study (around 0.2 blue cod per minute per rod) (Figure 2). Control sites ranged from 1.3 to 5.6km from the reserve boundary, suggesting there was no enhancement of CPUE by spillover at these distances for this species.
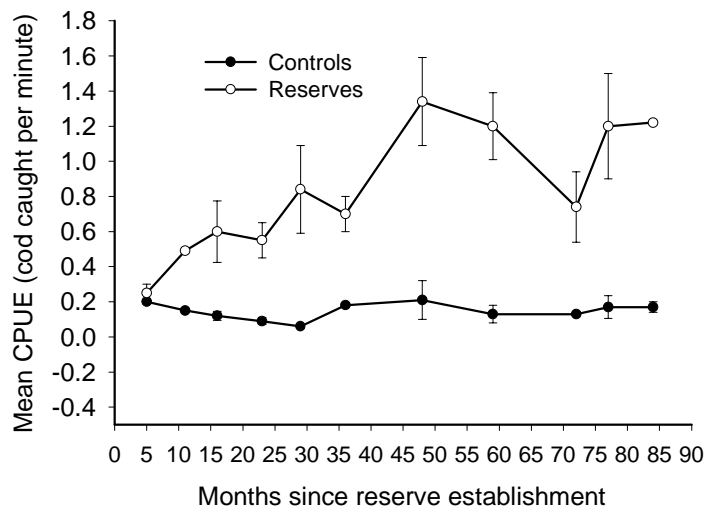
Figure 2: Mean catch per unit effort in blue cod per minute fished per rod for reserves and control sites from September 1993 to April 2000. Error bars show +/- 1 S.E. Redrawn with permission from Davidson 2001.

Davidson's (2001) study provides the longest time series of data available for any New Zealand Marine Reserve (9 years for densities and 6.5 years for fish lengths and catch rates). It demonstrates dramatic changes in the size, population structure, abundance and behaviour of the dominant reef species, blue cod. In April 2000 (after 84 months of protection) blue cod were over 125% more abundant within the reserve and on average 8cm longer than in exploited sites. (The fish were bigger in the marine reserves than in the control sites at the outset of formal reserve management. Davidson attributes this to the previously voluntary closure of the area for 4 years by local dive clubs). A key finding from Davidson's (2001) research was that by the end of the study large fish were much more common in the reserve. Thirty five percent of cod in reserves were larger than 33cm compared to <1% in fishing grounds. Since fecundity of fish increases exponentially with body size, this size difference means that egg production from the protected population will likely be many times greater than from fished stocks.

Key points

- New Zealand marine reserves were primarily established for conservation purposes but have demonstrated the potential to benefit fisheries for major commercial and recreational reef species.
- Lobster and fish populations have built up rapidly, increasing in abundance and body size in small marine reserves (around 400ha) and adult movements make them available to nearby fisheries.
- Catch per unit effort of lobster fishers around the Leigh Marine Reserve was equivalent to that in unprotected areas, despite nearshore reefs being closed to fishing.

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4. The effects of marine reserves in South Africa on commercial and recreational fisheries

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The development of marine reserves in South Africa, which has some of the largest no-take marine reserves in the world, was partly attributable to the country's long history of terrestrial protected area management. Large areas of land were set aside for conservation purposes and the rationale behind terrestrial protected areas extended naturally to marine protected areas. Very few of South Africa's marine reserves were established for fisheries management but the positive effects that they were having on fisheries soon became evident.

South Africa has incredibly diverse and productive marine fisheries. Reported annual landings exceed one million tons, but actual landings are considerably greater. The full transition from cold-temperate to warm subtropical waters along South Africa's 300 km coastline results in the high diversity of fisheries species, and the high productivity is due to the world second largest upwelling system, the Benguela Current.

The large demersal and pelagic fisheries are well managed, with tightly controlled quotas and rights allocations. Stock assessment procedures for offshore commercial fisheries have benefited from a reliable time-series of fish catches and survey data. In contrast, inshore fisheries have been more difficult to manage. Rampant increases in inshore fishing capacity and illegal fishing have seriously reduced stocks of reef-fish and invertebrate species by recruitment over-fishing. Furthermore, the absence of reliable historical information has frustrated attempts at stock assessment for inshore species.

South Africa currently has 21 marine reserves, with more areas planned (Attwood et al. 1997, Mann 1998). A total of 4.9% of the coast is protected in no-take reserves and 19% of the coast is in protected areas of one form or other (Attwood et al. 1997). No data are available on the status of adjacent fisheries before these reserves were established, so it has not been possible to look at fishery effects over time, even though some of South Africa's marine reserves have been in place for over thirty years. The fisheries have also developed so much over the past few decades that such comparisons would be of questionable use. A consistent feature of inshore fisheries (commercial, recreational and subsistence) has been a steady increase in effort over the course of the twentieth century. It has been very difficult to distinguish between the effects of protection, the increase in effort and the natural variability of the fisheries. Instead, much has been learnt from research into the response of fish populations to protection in South African marine reserves, and from inferences based on experiments on fish movement. Most work has been conducted at two sites, Tsitsikamma National Park and De Hoop Marine Reserve.

Tsitsikamma National Park

The Tsitsikamma National Park (TNP) was established in 1964. TNP covers a 60 km stretch of the south coast of South Africa, in the Eastern Cape province, and extends three nautical miles offshore. No fishing is allowed in the TNP, although prior to 1999 a 3 km stretch of TNP coast was open to shore-anglers to boost occupancy of the adjacent rest-camp accommodation. It was later discovered that anglers formed a minor component of tourists, and this concession was abolished.

TNP provides a refuge for many species of reef-dwelling seabreams (Sparidae), endemic to South Africa, and which form an important part of the recreational linefishery. This was the conclusion of a comparison of fish density and size, as assessed by underwater census, at comparable sites inside and outside the marine reserve. In a study of the abundance of three sparid species (*Chrysoblephus laticeps, C. cristiceps* and *Petrus rupestris*), Buxton and Smale (1989) and Buxton (1993) found that within the reserve fish were more abundant, faster growing and larger, than in exploited areas. Another important difference among the protogynous hermaphrodite (sex changing) species was the sex ratio at different sites. Males, which are the larger sex, were relatively more abundant at the protected site. Hermaphrodites changed sex at an earlier age at exploited sites that at the protected site. This, together with the growth rate difference shows how exploitation changes life history parameters of fish. The lesson from these findings is that the impact of fishing on the capacity of stocks to recover from fishing is greater than what can be deduced from abundance alone. A skewed sex ratio limits reproduction, while smaller female size sharply reduces the egg-laying capacity of the stock. Buxton (1993) concluded that marine reserves were necessary to maintain the spawning capacity.

Cowley et al. (2002) examined the effects of the TNP on fish abundance by comparing catch-per-unit-effort (CPUE) and fish size data obtained from a roving creel census of anglers along the south-eastern cape with

experimental data obtained in the TNP. The four species studied (three seabreams: *Diplodus sargus capensis*, *Diplodus cervinus hottentotus* and *Pachymetopon grande*, and one dichistid, *Dichistius capensis*) were 5-21 times more abundant in TNP compared to exploited areas, and their mean sizes were larger in the TNP.

Cowley (1999) and Cowley et al. (2002) studied movement patterns by tagging fish. All four of the species mentioned above were resident, showing a high degree of site-fidelity. Another species (the seabream *Lithognathus lithognathus*) was found to be resident only as juveniles – the adults undertake long spawning migrations. This pattern of resident juveniles and migratory adults is shared by a number of other seabream species, including *P. rupestris* (Brouwer 2000). TNP maintains the spawning capacity by providing an effective refuge for resident fish or resident stages of more mobile fish.

There has been much speculation about the movement of larval stages from the TNP to adjacent areas. Analyses of ocean currents in the TNP suggest that truly planktonic larvae spawned in TNP may have little chance of remaining there. Current speeds frequently exceed 50 $cm.s^{-1}$, even near the ocean floor (Tilney et al. 1996, Attwood et al. 2002). There is therefore great potential for enhancing exploited populations through increased larval settlement outside the protected area. However, there is no conclusive evidence of widespread larval dispersal. Indeed, an exhaustive search for the larvae of seabreams recorded surprisingly few, raising the possibility that seabream larvae might employ methods to avoid transport by currents.

The TNP forms part of the spawning area for chokka squid (*Loligo vulgaris reynaudii*), an important commercial species (in 1993 the fourth most important fishery in South Africa), and squid nests have been recorded at ten sites in the TNP (Sauer 1995). These are the only nests that are not disturbed by the squid fishery. Apart from removal of squid, the fishery disrupts the mating process, and anchors damage the egg-beds. Because individual chokka squid spawn at a number of nests, the TNP is unlikely to provide absolute protection for any part of the stock, but it undoubtedly reduces the fishing pressure and protects some spawning areas from disturbance. Overall there are 238 nesting areas along the south-eastern Cape, which means that the TNP protects just under 5% of the breeding sites.

Marine reserves in South Africa: the wider picture

The De Hoop Marine Reserve is 326 km west of Tsitsikamma. It was established in 1985 and protects a 50-km stretch of coast, extending three nautical miles from shore, and with a total area of 36,000ha. It has also been shown to protect populations of fish important to commercial and recreational fisheries, particularly surf-zone fish (Bennett and Attwood 1991, Attwood and Bennett 1994, Attwood and Hawkins in Roberts and Hawkins 2000).

Monitoring of fish density by experimental angling in the De Hoop Marine Reserve showed that the density of seven of ten species recovered after the reserve was proclaimed in 1985. The evidence for the others was inconclusive due to extremely infrequent or sporadic catches. Overall CPUE within the reserve from the experimental angling undertaken up to 1992 was an order of magnitude greater than that in adjacent exploited areas (Bennett and Attwood 1993).

Attwood (2002) studied one species, the galjoen (*Dichistius capensis*), in great detail and reached some conclusions that may be generally applicable to fisheries in South Africa and beyond. While there is little doubt that galjoen is more abundant in protected areas than elsewhere, there is no direct evidence that these areas have led to an increase in fishing yield for this species, despite the fact that adults and possibly larvae too, leave the protected areas and become available to the fishery. As in most other parts of Africa, the human population of the South African coast is expanding rapidly, and so too is the pressure on coastal resources. Over the course of the twentieth century, fish stocks have steadily declined as fishing effort has increased. Despite initial predictions that the marine reserves would markedly improve fishing, the reality of increasing pressure should have alerted scientists to a more realistic prognosis. A modeling investigation based on detailed tagging research, suggests that De Hoop, Tsitsikamma and a few other smaller marine reserves have served to retard, and in some areas halt, the decline in fishery yield. Indeed, surveys have not shown any consistent change in the rate at which galjoen have been caught over the last twenty years (Bennett 1991, Brouwer et al. 1997).

Modeling studies are indispensable in the investigation of reserve effects. It is difficult to prove that marine reserves arrested a decline in the fishery, as there is no possibility of repeating the experiment or running a control without marine reserves. Therefore, it is necessary to extrapolate from what is known to reach likely conclusions on the effect of a particular strategy. In this case, it was known that the stock of galjoen had declined sharply, that the reserves led to increases in abundance within those areas, that small yet substantial parts of the protected stocks moved out of the reserves to be caught elsewhere (Attwood and Bennett 1994) and that no

further declines in catch rate were detected. Whereas critics of the value of marine reserves cite the absence of any improvement in yield as a failure of this approach to management, fishers can be thankful that their catches did not decline further, as the original trend suggested they inevitably would. The models indicate that without reserves, catches would have collapsed (Attwood 2002).

Compliance and support

In a study of boat-based line fishers' attitudes to fisheries management regulations between 1994 and 1996 (Sauer et al. 1997), in the Eastern Cape area (which includes Tsitsikamma) 92% of the 96 commercial anglers interviewed agreed with the use of reserves and said they obeyed reserve regulations. For recreational anglers, 93% of the 118 fishers interviewed said they agreed with the use of reserves and 84% said they obeyed the regulations. Other fisheries regulations that fishers were asked about included size limits on fish, bag limits and closed seasons. Amongst commercial fishers in the Eastern Cape there was more support for reserves than for the other fisheries management techniques (Sauer et al. 1997).

Brouwer et al. (1997) made a similar survey of attitudes of recreational shore-anglers to different management approaches. Along the Eastern Cape coast more people agreed with size limits than with marine reserves, but 86% said they agreed with marine reserves and 91% said they complied with reserve regulations. Overall, 84.5% of South African line fishers participating in that survey supported the existing marine reserves. In a survey of South African spearfishers, Mann et al. (1997) found that more agreed with the use of marine reserves (88.9%) than with minimum sizes, bag limits and closed seasons. Only 13.2% of spearfishers admitted to fishing illegally in marine reserves.

In a review of marine reserves in South Africa, Attwood et al. (1997) assessed the problem of illegal fishing in reserves. In TNP, poaching of most commercial fish species in coastal waters (including galjoen, and all the seabreams) was considered a cause for concern. The poaching of a number of near-shore invertebrates (periwinkles, limpets and octopus) was also a concern. Increases in the biomass and CPUE of some of the important finfish species inside the TNP have therefore been achieved with imperfect compliance.

If correctly used, marine reserves hold another potentially valuable benefit for fisheries management. Sustainable fisheries management depends critically on the ability of the fisheries management agency to assess the state of the fish stock in relation to some threshold or reference values. Typically, this involves examining catch and effort trends, but because few fisheries have time-series that began at the inception of the fishery there is usually a lack of comparative values. Consequently contemporary measurements carry very little information for decision makers. South Africa's marine reserves have been used to provide reference points and estimate the parameters of the unfished stock that were not available from historical information. It is only by comparison of exploited and unexploited areas, that the mortality rate could be separated into a natural and a fishing induced component, a vital parameter for any fishery model. Furthermore, the cheapest form of fishery monitoring, the gathering of catch per unit effort data, provides a useful indicator of the state of the stock when compared to experimental data obtained from unexploited areas. South Africa is in the process of developing assessment procedures to utilize these comparisons more effectively. Marine reserves are thus an integral part of the fishery management procedure.

Key points

- Tsitsikamma Marine Park and the De Hoop Marine Park are long standing marine reserves that have been highly effective in protecting stocks of exploited species in South Africa.
- A small but substantial fraction of reef-fish leave reserves and contribute to the fishery.
- In all likelihood, marine reserves halted the decline and stabilized catches of inshore stocks that are being subjected to ever-increasing pressure from a growing coastal population. In view of this increasing pressure, it was unrealistic to expect fishing success to improve as a result of the introduction of reserves.
- Marine reserves in South Africa complement comprehensive fisheries management programmes using size limits, bag limits and gear restrictions.
- Marine reserves help in assessment of the state of exploited stocks, thus benefiting fisheries indirectly.
- Marine reserves are supported by the majority of fishers in South Africa.

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5. Lobster fisheries management in Bonavista Bay, Newfoundland, Canada

The Newfoundland lobster fishery has greatly increased in importance in recent years due to collapse of groundfish stocks and closure of the cod fishery. The fishery targets American lobster (*Homarus americanus*) and is conducted from small open boats close to shore. There was concern over the future of lobster landings in Canada because of high exploitation rates and the high proportion of immature lobsters being harvested, and in response to this the Canadian Fisheries Resources Conservation Council (FRCC) developed a conservation framework for the species (FRCC 1995, in Rowe 2002). On a local level, responding to FRCC recommendations, the Eastport Lobster Protection Committee was established in 1995 to work on management techniques to promote a sustainable fishery. The committee was formed by local fishers and supported by the majority of lobster fishers in the area. One of their main roles was to reinforce the importance of complying with regulations on capture sizes, gear regulations and releasing egg-bearing females. In 1996 a voluntary programme of marking and releasing egg-bearing females was introduced. Females with eggs were marked with a v-notch in their tail and when caught again, even if not bearing eggs, they would be released, so improving the opportunities for spawning individuals to continue to produce. There was good compliance with size and gear regulations amongst Eastport fishers and there were increases in landings.

In 1997 the Committee initiated protection of two areas of good lobster habitat within their fishing grounds, 2.1km^2 in total and supporting an estimated 1.5% of the lobsters in local fishing grounds. They also restricted lobster fishing to traditional users (about 50 licensed lobster fishers). In return, traditional users gave up the right to trap outside their area. There was also a buffer zone where fishers from Eastport and elsewhere could fish. Users were thus limited to fishing in their local area, assisting in the management of the resource. These two management decisions were the main impetus behind the establishment of the Eastport Peninsula Lobster Management Area which incorporated the two closed areas (Rowe and Feltham 2000).

The whole process of establishing the closed areas was internally motivated. The committee became aware of the possibility of using area closures to manage their lobster resources through the FRCC report. The management plan, including selection of the protected sites, was crafted using local knowledge integrated with scientific advice. The committee also recognized the need to monitor the effectiveness of their management initiatives and applied for funding to monitor populations with the assistance of graduate students from the Memorial University of Newfoundland. Fishers carried out experimental fishing and participated in tagging studies and much of the data was analysed at the University and results fed back to the committee and the fishing community. Members of the community (fishers and their families) and a local school have also participated in data analyses. This helped increase awareness of the importance of conserving stocks and included the wider community in the management effort. The closed areas are policed by the lobster fishers themselves in a system of peer enforcement which has been very effective (Rowe and Feltham 2000).

Biological evidence for the effectiveness of closed areas is growing. In a study of lobster populations between 1997 and 1999, Rowe (2001, 2002) found a low frequency of lobster emigration from closed areas, offering increased survival to protected lobsters. Over 90% of tagged lobsters were recaptured in the same area they had been tagged in. In a study of lobster populations inside and outside closed areas, in one of the two areas (Round Island) lobster density, sizes of individuals, and proportion of egg-bearing females were significantly greater than that in adjacent fished areas. Lobsters were significantly larger inside the other closed area (Duck Island), but there was no significant difference from fishing grounds in density or proportion of egg-bearing females. These studies were undertaken only three fishing seasons after the reserves were established. Fishing pressure outside closed areas was intense with mortality estimated at nearly 72% of lobsters of harvestable size and condition (Rowe 2001). Rowe concluded that closures were already demonstrating the potential to benefit local fisheries, even after just 2-4 years of protection. Larger lobsters and more egg-bearing females in the closed areas will increase egg production. Only a very small proportion of lobsters within the wider management area are currently protected, but at present egg production is thought to be at such low levels that even this small number of lobsters producing significantly more eggs could have a positive effect on future numbers of lobsters in the area. Egg production is estimated to have increased (Ennis 2000) and closed areas are thought to have contributed 7.1% of total annual egg production.

Eastport lobster fishers have reported improved landings over the management period, whereas in the rest of Newfoundland fishers have seen a serious decline in their landings (Figure 1). In the Eastport Peninsula Lobster Management Area lobster populations appear to be recovering in response to the package of management techniques introduced, the most important being the two closed areas and the marking and returning of egg-bearing females. The success of the programme has prompted other communities in Newfoundland to consider similar initiatives. It has also prompted the Eastport community to consider totally closing the lobster closures to fishing, establishing them as marine reserves to protect other species as well.
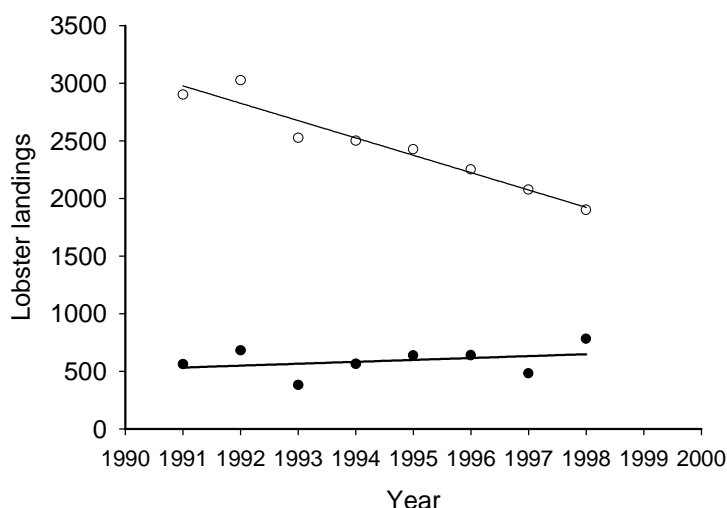
Figure 1: Lobster landings for Newfoundland (metric tonnes), excluding Bonavista Bay, and lobster landings for the Eastport Peninsula, 1991-1998. Redrawn from Rowe and Feltham 2000.

This study emphasises the success which can be achieved through an initiative that originated and developed in the community. The closed areas of the Eastport Lobster Management Area are unusual because they were established by fishers for fisheries management. The effectiveness of peer enforcement and the usefulness of involving the wider community in designing, enforcing and monitoring the closed areas seem to be the basis for continuing success. The programme was initiated and developed into the success it is today by one lobster fisher in particular, Mr George Feltham. In 2000 he received the Canadian National Award for Sustainable Fishing in recognition of his vision in establishing the management committee and the community based management and monitoring programmes.

Key points

- Management measures, including closed areas, were initiated by lobster fishers themselves.
- Loss of fishing grounds was divided evenly amongst the fishers so nobody was hit too badly.
- Lobsters in closed areas are increasing in abundance, living longer, growing larger and producing more eggs than those in fishing grounds, potentially helping contribute to improvement of local stocks and supporting fisheries through increased recruitment and spillover.
- This pilot study shows that the reserves work, but as they are very small (in this case protecting an estimated 1.5% of the population), more will be needed to fulfill their full potential to boost fisheries. Fishers are now interested in having a larger no-take area.

With thanks to Sherrylynn Rowe.

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6. Georges Bank fishery closures, USA

Georges Bank is in the Gulf of Maine on the eastern seaboard of the United States and was once one of the most productive fishing grounds in the world (Kurlansky 1997). However, decades of increasingly intensive fishing led to a series of fishery declines and collapses that precipitated a drastic new management initiative. In 1994, three large areas totalling 17,000km^2 were closed to fishing for groundfish (Figure 1), a multispecies group of bottom living fish including cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), yellowtail flounder (*Limanda ferruginea*), witch flounder (*Glyptocephalus cynoglossus*) and silver hake (*Merluccius bilinearis*), among others. This represents approximately 25% of fishing grounds on the bank.
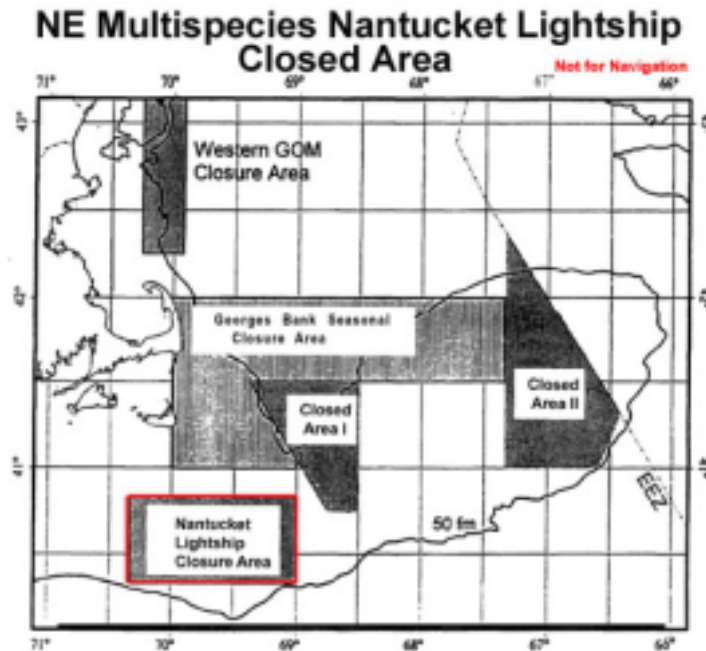
Figure 1: Map showing closed areas in the Gulf of Maine. Closed Areas I and II, and the Nantucket Lightship Closure Area were protected in 1994. Figure courtesy of the New England Fishery Management Council.

These areas were also closed to all kinds of fishing gear that might catch groundfish incidentally, or damage their habitats, such as scallop dredges. An additional purpose of the closures was to help rebuild depleted scallop stocks (A. Rosenberg, pers. comm.). Although the areas were not closed to all forms of fishing (long-lining was permitted, for example), and so do not qualify as fully protected, they offer many important insights into how large, fully protected marine reserves might work. They also show how it is possible to create and police large, offshore areas that are exploited by industrial scale fisheries. In this case, a substantial fraction of the vessels using the bank were equipped with satellite monitoring systems that collected information on their location at regular intervals. The data were transmitted to a monitoring centre and used by the National Marine Fisheries Service to verify compliance with the boundaries of the closed areas, and enforce regulations. Satellite tracking showed that there were very high levels of compliance (Murawski et al. 2000, Rago and McSherry 2002). In 2002, courts secured the first conviction using satellite data alone of a vessel fishing illegally within one of the closed areas.

Murawski et al. (2000) judged the closed areas as a major success. They significantly reduced fishing mortality of depleted groundfish stocks. Stocks of haddock, yellowtail and witch flounders in particular have been on the increase. While many of the key species still lie well below historical levels of biomass the trends have turned upwards after many years of decline. Yellowtail flounder is now gradually approaching target biomass levels for maximum sustainable yield (Cadrin 2000). Cod have been slower to respond, perhaps because they are more mobile than haddock and flounders, but there are encouraging signs that their biomass is rebuilding (NEFSC 2001, O'Brian and Munroe 2001). Unpublished experimental trawl data from the Northeast Fisheries Science Center demonstrate spillover of several species of groundfish from closed areas to fishing grounds (Paul Howard, New England Fishery Management Council, pers. comm.). The effects are beginning to be felt by

fishers. At a meeting of fishers and scientists in Oregon in 2001, a Cape Cod fisherman described how prior to the closures, he had to steam 70 miles one way for a day's catch of around 270kg of cod, but these days he travels just 30 miles toward the edge of the closure and catches 500kg.

These improvements in fish stocks were not gained using the closed areas alone. At the time of their implementation in 1994, a package of other measures was introduced which limited the number of permits to fish for groundfish, increased trawl mesh size by 1/2 inch, and initiated a stepped effort reduction program aimed at halving time spent fishing for groundfish by 50% over 5 years. The latter timeline was speeded up when the depth of the cod collapse became clearer. This combination of conventional fishery management measures and large closed areas has evidently been highly successful and represents a model for others to follow.

The most dramatic effect of the closures was on scallops (*Placopecten magellanicus*). By 1994 scallops had been heavily depleted by fishing. They rebounded after five years of protection, reaching 9 to 14 times the density of legal size scallops in fished areas (Murawski et al. 2000). Satellite monitoring of fishing vessels showed scallop fishers hugging the edge of the closed areas, benefiting from high catches as a result of export of scallop offspring on ocean currents (Rago and McSherry 2002). This is exactly what theoretical studies suggest would happen. Biophysical modeling of current patterns across Georges Bank indicate that the closed areas export scallop larvae to large regions of the bank as a result of their 40-day pelagic larval duration and the prevailing anticyclonic circulation. The models also suggest that the closures are able to resupply themselves, with substantial self-recruitment likely (Lewis 1999). Furthermore, patterns of fishing effort revealed by satellite monitoring show effort concentrated into places most likely to receive the greatest volume of larval supply (Lewis et al. 2001, Rago and McSherry 2002).

In 1999 the southern part of Closed Area II was controversially reopened to fishing for scallops to allow fishers to extract some of the accumulated adult biomass but with bycatch limits imposed for yellowtail flounder. This has allowed some of the largest catches to be taken for at least a decade. Many see this as the first step toward management of scallop fisheries by rotating closed areas, a strategy that some models suggest would be useful for a species with its life-history (Quinn et al. 1993). However, such rotational management should complement rather than replace permanent closed areas, as reopenings will undo some of the beneficial effects of protection, such as improvements in habitat.

Protection of closed areas from mobile fishing gears has led to changes in habitats on the bank. Five years of protection from trawling have led to large increases in the density, biomass, species richness, and production of bottom-living animals relative to areas open to trawling (NRC 2002, J. Collie pers. comm.). In turn, these effects can be expected to enhance production of commercial fishery species (Lindholm et al. 1999, Roberts and Sargant in press), leading to long-term sustained benefits from protected area management.

Key points

- Large-scale fishery closures on Georges Bank show that reserves can be effective for industrial fisheries in temperate continental shelf settings.
- Satellite technology and vessel monitoring systems make it possible to police and verify compliance with large offshore closures.
- In five years of protection, closures have reduced fishing mortality and begun rebuilding stocks of depleted groundfish.
- Scallops responded quickly to protection, and adjacent fisheries have been enhanced by export of their larvae.
- Scallop fishers fish-the-line along the borders of closed areas, benefiting from higher catch rates there.

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7. Marine parks and other protected areas in the United States Virgin Islands

The Virgin Islands National Park was one of the first marine protected areas to be established in an American territory. The park was established on St. John, in the US Virgin Islands in 1956, and in 1962 it was extended to include 2,287ha of the adjacent waters. At the time the park was established, fishing effort in the area was low and mainly artisanal, using traditionally designed traps. Traditional trap fishing was permitted in the park because it was not felt that this had an impact on fish stocks, but spearfishing was banned and catches of conch (*Strombus gigas*), whelk (*Cittarium pica*), and spiny lobster (*Panulirus argus*) were restricted. Over time expanded commercial fishing and more efficient gears replaced the traditional small-scale trap fishery in the park. Initially traps had a mangrove wood frame and vine or chicken wire mesh. Later fishers began to use traps constructed of steel and plastic-coated wire.

Garrison et al. (1998) surveyed commercial fish traps underwater to assess the status of the trap fisheries inside and outside the Virgin Islands National Park between 1992 and 1994 (after 36-38 years of protection). They found that there were significantly more groupers and parrotfish per trap inside the park compared to outside, but for groupers they believed this to be a sampling artifact because so few were caught. Numbers of surgeonfish and parrotfish per trap were also higher inside the park than outside. However, numbers of commercial size parrotfish and triggerfish per trap were higher outside the park than inside. Garrison et al. (1998) concluded that the level of protection offered by the park to fisheries species was insufficient. The intensity of trap fishing in the national park meant that the park status provided virtually nothing in terms of species or habitat protection, and catches in the park indicated that the area was heavily overfished. From 1992 to 1994 the number of fish per trap and the size of the fish decreased, and there was also a trend towards catches dominated by herbivorous species like surgeonfish, and less of the preferred commercial species such as groupers, snappers, bream and triggerfish. When the catch rates inside and outside the marine park were compared with other areas in the Caribbean and Florida, they were found to be low and the proportion of herbivores high. This study also revealed the importance of protecting the whole range of habitats, not just coral reefs. Garrison et al. (1998) found that 78% of fish observed in their surveys were captured in adjacent non-reef habitats such as algal plains and gorgonian forests.

Wolff et al. (1999) investigated trap catch rates in the Virgin Islands National Park and at another site on St. John, and confirmed that catch rates were higher in gorgonian-dominated habitats than on nearby coral reefs. They found the numbers of fish observed on coral reefs were significantly higher than in gorgonian habitats, but that the mean catch per trap haul from gorgonian habitats was nearly twice that from reefs. Parrotfishes, surgeonfishes, squirrelfishes, groupers and butterflyfishes were all significantly more abundant in trap catches from among gorgonians. Wolff et al. (1999) also found that effective area fished by each trap was higher in the gorgonian habitat than on coral reefs. Because coral reefs had higher rugosity, Wolff et al. (1999) suggest that traps in gorgonian habitat offer structure for fish, and therefore are more attractive to fish in this habitat than to fish on reefs. Trapping in gorgonian areas, which often fringe reef areas, may also take advantage of the daily migratory behaviour of many fish moving from resting places on reefs to foraging areas in adjacent habitats. This work emphasises the importance of protecting the full range of habitats from fishing in order to ensure effective protection of reef fish populations.

Rogers and Beets (2001) reviewed the effectiveness of the Virgin Islands National Park and also the Buck Island National Reef Monument on the nearby island of St. Croix (established in 1961). They synthesized a large body of work on marine resources in these parks conducted over decades. Stresses such as boat damage, hurricane damage and coral disease are having serious impacts on habitats and species within the national parks, as they are across the Caribbean region. Over the years these authors have seen coral cover decline and cover of macro-algae increase in protected areas. Seagrass cover has decreased as a result of anchor damage and storms. Populations of important commercial predatory fish such as groupers and snappers have greatly decreased and herbivorous fish have increased as a proportion of the fish community. The regulations that allow "traditional trap fishing" now mean that intense levels of commercial trap fishing go on inside the marine parks. As a result, the parks offer no protection to the majority of reef fish species vulnerable to capture by traps and Rogers and Beets (2001) could detect no difference in the numbers of fish caught per trap, the biomass of fish estimated on underwater surveys or the mean size of fishes between areas inside and outside the Virgin Islands National Park.

Rogers and Beets (2001) attribute the failure of these national parks to protect the habitats and species to a combination of natural impacts such as those from hurricanes and disease, and human impacts such as fishing and boat damage. The lack of protection offered by the regulations is a major contributor to their failure and even these limited regulations are poorly enforced. They recommend more active protection in the form of no-take marine reserves, seeing it as critical for the recovery of habitats and species.

Elsewhere in the US Virgin Islands a protected area has been more effective in protecting fishery species and allowing populations to recover. In 1990 a grouper spawning aggregation seasonal closure was introduced off the island of St. Thomas, covering 1.5% of the fishing grounds for groupers (Bohnsack 2000). During the 1970s and 80s grouper spawning aggregations were targeted by fishers and at least two were completely eliminated by fishing. Catches of red hind (*Epinephelus guttatus*) were declining, and the Nassau grouper (*Epinephelus striatus*), which used the same aggregation sites and was once the main grouper in fish catches, became extremely rare. There were fears that the red hind fishery would collapse in the same way. After seven years of protection of the spawning aggregation, Beets and Friedlander (1999) found that the average length of red hind had increased by a third from 29.5cm (the mean length in 1988 before protection) to 39.5cm. The sex ratio had also changed from 15 females per male in 1988 (this skew being a result of the selection of the larger males by the fishery) to 4 females per male in 1997. A better ratio of males to females is expected to facilitate greater reproductive success, and enhance the population of this important fishery species. In 1998 this seasonal spawning aggregation closure became a fully protected permanent marine reserve (Beets and Friedlander 1999).

Key points

- If the protection of marine parks is limited to gear restrictions and species restrictions that fail to address the major extractive used of the area, protection is unlikely to be effective, especially when fishing pressure is high or increases over time.
- It may be necessary to protect the range of habitats used by species to ensure effective protection.
- Protecting spawning aggregations of key fishery species can be highly effective in averting the decline and commercial extinction of vulnerable species like groupers.
- Even small marine reserves can offer effective protection to migratory species if they are located where such species are highly vulnerable to targeted fishing effort.

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8. Community-based closed areas in Fiji

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Fiji has a very strong traditional resource management system. Communities own and manage specific areas, and there is local autonomy in decision making about use of resources (Aalbersberg et al. 1999). In the early 1990s, in response to the now familiar story of a community coming to terms with declines in marine resources and consequent decreasing catches (in this case of marine invertebrates), the residents of Ucunivanua village in the Verata district of Fiji decided they had to deal with these problems. In consultation with the University of the South Pacific and the Biodiversity Conservation Network they put in place several management strategies. These included replanting mangroves, banning mangrove cutting, coral extraction and fishing using poisons, obtaining alternative income from a bioprospecting enterprise, and setting up a species-specific fishery closure.

The purpose of the closed area or species refuge was to act as a replenishment zone where over-exploited species would have a chance to recover, and increase harvests in adjacent fishing areas by spillover. The species initially managed by this closed area was the clam (*Anadara* sp.). In 1997 the community designated a 24ha area of seagrass and mudflat directly in front of the village and closed it to clam harvesting. The community knows the closed areas by local names and understands the boundaries, which are also marked using stakes and other markers. By implementing this refuge, the village was returning to a traditional management practice of *tabu* areas, temporarily closed to fishing.

An important component of this project was the involvement of the Ucunivanua community in monitoring the effects of the closed area. With the support of the University of the South Pacific, the community have been assessing changes in abundance and size of clams in their closed area and adjacent fishing grounds. University scientists trained community members in sampling techniques, using transects and quadrats, and measuring the clams encountered. It took about two weeks to train the team to collect data themselves.

Figure 1: Ucunivanua monitoring team counting and measuring clams found inside quadrats along their transect line in the intertidal *tabu* area. Photo by John Parks.

Community monitoring is working well. People are really enthusiastic about the work and keen to see results of their management. Tawake and Aalbersberg (2001) of the University of the South Pacific did a parallel study of invertebrate numbers at the same sites as community researchers were assessing. They found no significant differences in the data obtained, confirming that the community monitoring was of the same quality as a scientific study conducted by outside experts. However, university scientists assisted with data analyses which the community found more difficult. The scientists felt an important part of their contribution was regular return visits to the communities to follow up training, monitoring and data analyses. The local monitoring team and the university scientists make regular reports to village and district meetings, so that the results can be used in future management decisions. Local monitoring of reserve effects contributes greatly to the autonomy of the management area, and is also much more likely to continue in the long term if it becomes part of the routine within the community. Communities are also more likely to respond to monitoring results they have collected themselves and therefore trust, rather than when they are only shown the findings of studies conducted by external researchers. Community monitoring encouraged people to improve their compliance with the closed area, and initial compliance problems diminished and the community were also motivated to protect the area from poachers. The location of the closed area directly in front of the village allows it to be policed more effectively.

Monitoring showed a dramatic increase in the numbers and size of clams in the closed area after three years of protection, and an increase in the numbers of smaller clams recruiting to fished areas (Figure 2). Clams began to grow bigger than had been seen for three generations. People had hardly ever seen clams of more than 6cm, but began to find individuals up to 9cm. After four years of management, clams had increased in abundance by 1353% in the closed area and by 523% in the fished area. Catch per unit effort increased and people only needed to spend half the time they normally spent collecting clams to get the same catch as before (Tawake and Aalbersberg 2001). The most recent data collected in 2002 shows that after 5 years of protection there have been further increases in clam abundance. Clams are now 1796% more abundant in the closed area and 719% more abundant in the fished area (Ucunivanua community and A. Tawake unpublished data). Because of their sedentary lifestyle and their pelagic larval stage, molluscs like clams might be expected to respond particularly well to management using closed areas.
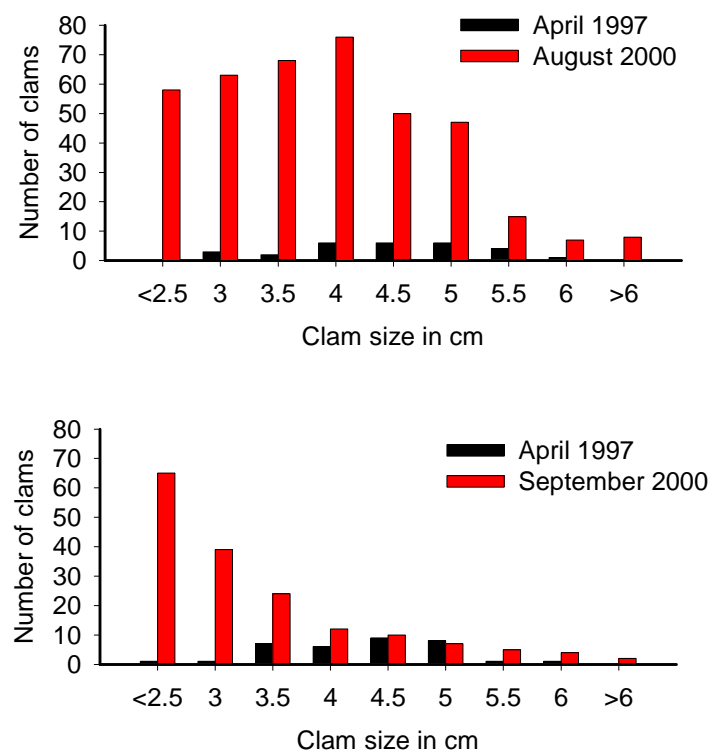
Figure 2: The number of clams (*Anadara* sp.) counted in 50 1m² quadrats in each size class in the closed area (top) and in the adjacent fished area (bottom) in 1997 when the closed area was established, and in 2000 after 3 years of protection. Data collected by the Ucunivanua Monitoring Team and reproduced with their permission. From Tawake et al. 2001.

People are also seeing some species that had become very rare in the area returning as their populations recover. Stingrays have become more common, perhaps in response to increases in populations of their prey species, including clams. Seahares (*Dolabella auricularia*), which are a local delicacy, have reappeared after people hadn't seen them for generations. They have also reported habitat improvements in the protected area, with seagrass and algae recolonising bare mudflats (perhaps encouraging the return of herbivores like seahares). It is likely that in addition to the species closure, the whole package of management strategies has contributed to this ecosystem recovery, for example the conservation of mangrove areas and the reduction in use of damaging fishing techniques.

Overall, the community has seen a 59% increase in household income, in part due to increased sales of clams and in part from income generated by the bioprospecting element of the project (Tawake and Aalbersberg 2001, LMMA fact sheet). Other benefits included increased community cohesion as the community worked together to manage their resources and monitor the effects. There was also the chance for resource users to influence provincial environmental policy and community and cultural pride increased as the initiatives have proved to be successful and popular with other communities. The initiative has increased awareness of environmental issues and the active participation of people in management and monitoring has generated interest in science and traditional management techniques.

Through representatives of Ucunivanua community speaking at meetings and through features in the media, the idea of using closed areas to allow stocks to recover and to enhance adjacent fisheries has spread throughout Fiji. It expanded from two sites with a total area of 0.63km^2 in two villages in 1997 to nine sites with an area of more than 7km^2 in seven villages in 2000. In Verata district there are five sites which now have monitoring data for between 1 and 4 years of management, all showing dramatic increases in the abundance of invertebrates in closed areas and adjacent fished areas. For example, at Kumi village after 2 years of management, clam numbers increased by 643% in the closed area and by 347% in the open area. At Sawa village, mudlobster (*Thallasina anomala*) numbers increased 5-fold inside the closed area and doubled in fished areas, also after 2 years of protection. There are now closed area projects going on in four other districts in Fiji and the pioneer team members of Verata district are involved in training community members in the new projects.

One of the problems with the single-species closures was that people collecting other species (most collectors targeted a number of species) were tempted by the forbidden species and sometimes took some illegally. This is one of the reasons that communities are now implementing completely closed areas. The Ucunivanua area has been permanently closed, but at the other villages the areas are closed temporarily so far, with the decision to open them resting with the chief on the recommendation of the community monitoring team. The success of closed areas on mudflats where they protected clam populations has also prompted communities to establish closed areas in mangrove and coral reef habitats to manage resources exploited in these habitats. Thus, a wide variety of habitats and species are now being protected by the *tabu* areas, and hopefully in future community-gathered data will be available on the effects of these broader closures on populations and catches of target species, and on the ecosystems in general. As well as using fully no-take areas, the *tabu* projects are developing to incorporate more detailed community monitoring, including socio-economic monitoring. On the basis of these success stories, the Fijian government has developed a programme of locally managed marine areas (LMMAs).

Key points

- Species closures have led to large increases in clam numbers inside closed areas and in nearby fishing areas.
- Catch per unit effort has increased adjacent to a closed area at Ucunivanua, and people only need to spend half the time fishing that they did before the area was closed.
- Species that had disappeared locally have returned, and habitats have improved in the closed areas.
- Communities have chosen to adopt fully no-take areas instead of single species closures because of the temptation of fishers to take restricted species while collecting other species.
- A small village project has become the catalyst for the use of similar systems throughout the country.
- One reason that closed areas have worked so well in Fiji is the strong tradition of community resource ownership and management and the ability of communities to make decisions about local management.

We thank Ratu Pio Radikedike (the community monitoring leader), the residents of Ucunivanua village and the Veratou project for sharing their data in this case study.

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9. The Sambos Ecological Reserve, Florida Keys National Marine Sanctuary, USA

The Sambos Ecological Reserve covers 9 square nautical miles of backreef and lagoon habitat. It is the largest of 23 no-take areas established as part of the Florida Keys National Marine Sanctuary (FKNMS), and is known as an Ecological Reserve rather than a Sanctuary Preservation Area (which the other 22 areas are). The Sanctuary Preservation Areas were designed to protect habitats, but also to reduce conflicts between user groups, whereas the Sambos Ecological Reserve was designed to protect an intact ecosystem (Leeworthy 2001). The socio-economic effects of the FKNMS are being monitored by a socio-economic monitoring programme which was designed by 50 social scientists and community stakeholders. The main goal is to "provide the knowledge necessary to make informed decisions about protecting the biological and natural ecosystems of the Sanctuary and its resources" (Leeworthy 2001). Two additional large Ecological Reserves were added to the network in 2001in the Dry Tortugas region.

The Sambos reserve became a no-take area in July 1997. Leeworthy (2001) summarises the finding of Thomas J. Murray and Associates (2000) who are assessing the financial performance of the reserve. They looked at the fishery catch value in 1997/8 and in 1998/99 to investigate whether fishers displaced by the reserve had suffered financially as a result of its creation, as had been anticipated in the initial management plan. They compared the earnings of Sambos fishers (who had been displaced by the reserve) with those of other fishers who fished nearby. They found that fishers' fears were unfounded. Net earnings actually increased after implementation of the reserve. The income of the Sambos fishers increased by 67% compared to an increase of 22% experienced by other fishers. Possible contributing factors to this apparent increase in earnings included the effect of Hurricane George, which led to loss of gear in 1998, and therefore lower earnings over the first study period. There was also a lobster trap reduction programme in the vicinity of the Sambos reserve which led to reduced costs per unit of effort and therefore increased earnings. The trap reduction programme formed part of the integrated management of the FKNMS, and was therefore considered as part of the package that included the no-take area. Overall, Leeworthy (2001) concluded that commercial fishers displaced from the Sambos reserve did not suffer any economic losses over the first two years of management, and actually saw improvements in the earnings that exceeded those experienced by other local fishers (Leeworthy 2001).

Suman et al. (1999) conducted interviews with fishers and other stakeholders involved in the 1995/6 consultation process on establishment of no-take zones in the Florida Keys National Marine Sanctuary (no-take areas were formally implemented in 1997). They found that commercial fishers felt excluded from the decision-making process and had concerns about the economic losses they believed they were facing due to the loss of fishing grounds. They were also concerned about the establishment of the no-take zones increasing recreational use, leading to crowding and conflict with their fishing activities. The fishers felt that NOAA (the US National Oceanic and Atmospheric Administration), the agency implementing the reserves, did not provide enough information about the possible negative impacts of the no-take zones, and that they had little say in their location. NOAA worked hard to provide information and to maximise participation to produce a zoning strategy that would be acceptable to resource users, and fulfil ecological criteria. However, fishers felt that their reports were not very user friendly and that NOAA did not listen to their points of view.

Spiny lobsters and stone crabs constitute over two thirds of the value of all commercial fish landings in the Keys (Murray and Associates 2000). Bohnsack and Ault (2002) present data on lobster traps in the vicinity of the Sambos Ecological Reserve in 2000/1 showing much higher densities of traps close to the boundaries of the reserve, compared to the surrounding area. This implies that fishers setting their traps around the reserve are experiencing benefits from spillover from the no-take area. The lobster fishery Sentinel Fisherman Program confirms this view (http://www.fknms.nos.noaa.gov/research_monitoring/Monitoring_Report_2000.pdf). It involves a local lobster fisherman who lays traps in the Sambos Ecological Reserve, on the edge of the reserve, and at control sites in open access areas. The program has found significantly higher catch-per-unit-effort in Sambos Ecological Reserve than in control sites, higher CPUE surrounding the reserve than at control sites, and larger lobsters in the reserve. Involvement of a fisher in monitoring has greater credibility with other fishers, which improves support from users and makes them more likely to comply with protection.

Annual monitoring of fish densities in no-take areas in the Florida Keys compared to fished areas showed increases in the densities of yellowtail snapper (*Ocyurus chrysurus*) in the no-take areas beginning in 1998 after just one year of protection, whilst in the fished areas densities of yellowtail decreased. For groupers (Serranidae), densities increased inside and outside the no-take areas, but this increase was faster in the no-take areas (Bohnsack and Ault 2002).

Key points

- The Sambos reserve is in its early days but there has been no loss of income to fishers as had been anticipated.
- Fishers are already concentrating trap fishing effort for lobsters around the boundary of the Sambos reserve and are experiencing greater catch per unit effort there.
- More generally, no-take zones in the Florida Keys have experienced rapid increases in stocks of commercially important snappers and groupers since their protection in 1997.

Thanks to Bob Leeworthy and Jim Bohnsack.

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10. The Nosy Atafana Marine Park, northeast Madagascar

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The Nosy Atafana Marine Park in Madagascar was created as a marine extension of a terrestrial national park in 1989, which has since become the UNESCO Biosphere Reserve of Mananara-Nord. The Biosphere Reserve has a total area of 140,000ha. Of this 22,649 ha is populated by 47,000 people. The marine park is a circular area with a diameter of 4km, covering approximately 1,000ha. The park is zoned into a no-take area of 20ha where fishing is not permitted and a buffer zone where regulated fishing is allowed. Within a relatively small area the Nosy Atafana Marine Park contains a wide range of marine and coastal habitats including mangroves, lagoons, seagrass beds, coral reefs, sandy beaches and coastal forests. There are four villages associated with the Nosy Atafana Marine Park, with a total population of 2275, of which 182 are active fishers (ANGAP 2002). The main objective of the Biosphere Reserve is sustainable development of the region and improvement of the quality of life of the coastal people, whilst conserving species and ecosystems and managing marine resources.

In the buffer zone fishers are permitted to fish on three specified days per week and only fishers from the four local villages are permitted to fish within the Marine Park. One of the most important management approaches contributing to success of the marine park was that in 1991 the Mananara Biosphere Project transferred property rights to the resource users through a contract. The contract lays out regulations regarding use of the area and its resources, specifies who is authorised to fish where, what the fishing gear restrictions and species restrictions are (for example, lobster collection is banned in the park), which activities are prohibited and what are the responsibilities of resource users. The contract can be adapted to circumstance and has resulted in users contributing actively to management, with their own initiatives for conservation and sustainable resource use. For example, the management committee looked into how damage caused by the harpoons used by octopus fishers may be reduced by use of octopus traps. In the future the contract needs to be incorporated into legislation.

There is a high level of community and stakeholder participation in management. Decisions about management are made through stakeholder sub-committees. Within the biosphere reserve there are six regional subcommittees, representing a wide cross-section of the community, which meet 2-3 times per year to assess management of the reserve. Education and training are very important in the management process. Through the management committees and the users' contract, park management is flexible and able to adapt. For example, when fishers were no longer permitted to camp in a traditional campsite, an agreement was made to allow fishers stranded by bad weather to stay in the park rangers' accommodation.

Figure 1: The resident park ranger records catches and inspections of the fishing gear are made to ensure they comply with regulations. Photo by E. Grandcourt

The marine park management falls under the responsibility of the director of the Mananara Biosphere Project. Two park rangers in a dugout canoe with a small engine patrol the marine park on the days when fishing is allowed, to ensure compliance with regulations. Catches from inside and outside the park are monitored, but the data have not yet been analyzed. Illegal fishing is punished with a ban on fishing in the park for between two weeks and three months depending on the offence. The project also has a fisheries section that monitors catches landed on the mainland adjacent to the park. The main fishing methods used are handlines for fish and harpoons for octopus. Records kept of illegal fishing in the park show low levels of recorded offences: 3-5 offences per year in 1998-2000 and 8-14 per year for using illegal gear.

The government of the Netherlands funded the marine park for five years up to 2001. The park also generates revenue from entrance fees for visitors and 50% of this is used for social development projects in the local community, for example sinking wells in villages without nearby access to water. However, the park is not currently financially sustainable and depends on outside funding. Increasing tourism in the area may contribute extra funds through user fees but tourism development needs to be managed carefully. Tourism is unlikely to become a significant source of funding in the near future because of lack of infrastructure in the region.

Evidence for the effectiveness of the Nosy Atafana Marine Park comes from the fishing community. They have observed an increase in reef fish recruitment within the park, and an increase in the numbers of adult reef fish in fishing grounds adjacent to the no-take zone. They have also seen an increase in catch rates inside the park in the buffer zone and in adjacent areas. In the surveys reported by Grandcourt et al. (2001), fishers and their fishing association representatives repeatedly reported on the increase in catch rates in and around the park. The use of fish aggregating devices has encouraged fishing outside the park and there have been opportunities for fishers and other marine resource users to diversify economically. Fishers whose activities were limited by rules imposed by the park could benefit from opportunities in agriculture, from which they could earn additional income and expand their economic possibilities. A loan system was also set up to help fishers to improve their fishing gear to allow them to fish in areas outside the park.

One potential threat to the marine park is illegal shrimp trawling by commercial vessels from urban areas. Although shrimping has yet to be reported within park boundaries, it does happen within the 2-mile exclusion zone of the adjacent coast. As the use of the marine park is restricted to members of the local community, some of the itinerant fishing effort has been displaced to other areas along the coast, some of which were already heavily exploited. This is viewed as one of the failures of the management system, and one that needs to be addressed in many coastal areas which are used regularly by itinerant fishers. Such fishers can come from nearby villages or from outside the local area, the country or the region. In this case, Grandcourt et al. (2001) suggest a wider, more integrated approach, with the Nosy Atafana Marine Park working together with other management initiatives along the coast.

Key points

- After more than ten years of effective protection of a no-take zone and a regulated fishing zone, fishers claim that fish are more abundant and catches have increased in the fishing zone and in nearby sites outside the managed area.
- Integration of stakeholders and the wider community with the planning and management process has led to success.
- The decision-making process is dynamic, with the flexibility to adapt changing management needs.
- Simple solutions are sought to management problems that are practical for the resource users to implement themselves, minimising the need for outside intervention.
- Allocation of property rights to fishers has encouraged community guardianship of the resources.
- More integration with projects in the region is necessary in future to deal with the issue of itinerant fishers, who through this project have only been displaced to other areas.
- The use of park revenues for important community projects such as building wells has brought direct social benefits to the wider community.

Thanks to Chantal Andrianarivo.

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11. Mombasa and Kisite Marine Parks, Kenya

Kenya has a long history of using marine reserves to protect coastal areas, promote tourism and more recently to manage fisheries (Obura 2001). At the moment marine protected areas account for about 5% of Kenya's nearshore area (Muthiga et al. 2000). The approach used most widely in Kenya has been to protect core zones from all fishing (marine parks) and to place regulated buffer zones where traditional fishing methods are permitted (marine reserves) around the parks (Obura 2001). Two marine parks have been particularly well studied since their establishment, the Mombasa Marine National Park and the Kisite Marine National Park. In general, reef fishery catches in Kenya, and elsewhere in East Africa, for example Tanzania, have declined over recent decades (Muthiga et al. 2000, McClanahan and Mangi 2001). Many poor coastal people in the region are dependent to some extent on nearshore coastal resources and their sustainable management is therefore very important. As tourism is an important part of the economy in Kenya, it is also necessary to manage potential conflicts between tourism and fisheries sectors.

The Mombasa Marine National Park

The Mombasa Marine National Park (MMNP) is in the south of Kenya and now has an area of 6.2km^2. It was established in 1987 as a no-take protected area and before its establishment the area was very heavily exploited. Some fishing continued in the park until 1991, and illegal fishing remains a problem (McClanahan and Kaunda-Arara 1996). When active management of the park began in 1991, 63% of the fishing grounds were designated as no-take areas and fishing effort decreased by an estimated 68% so fishing effort per unit area stayed approximately the same (McClanahan and Kaunda-Arara 1996). The available fishing grounds for fishers operating from the main local landing site, Kenyatta beach, decreased from 8km^2 to 3km^2, and the number of fishers reportedly decreased from 100 to 35.

Little information is available on the direct impact of the creation of the Mombasa Marine Park on fishers. However, according to McClanahan and Kaunda-Arara (1996) nearly a third of the fishers from the Kenyatta landing site either moved to fishing from another site or gave it up, suggesting that these fishers are likely to have suffered some serious negative effects. Through interviews with fishers, Rodwell (2001) found that one of the main impacts of the park on fishing costs was the increased cost of transport to the new landing sites that some displaced fishers moved to. In view of these problems it is not surprising that the early history of the park was turbulent. In 1995 conflicts associated with the closed area were tackled by a new management plan to reduce the size of the park and ban the use of beach seines. The park was reduced from an initial area of 8.2km^2 to 6.2km^2, by moving the northern boundary south by 1km and the southern border north (McClanahan and Mangi 2000). Since then the relationship between fishers and park management has improved (Glaesel 1997 in McClanahan and Mangi 2001).

The area to the south of the MMNP was established as the Mombasa Marine Reserve (MMR) where only those fishing methods considered traditional – traps, gill nets, handlines and octopus spears – were permitted. The area of the Mombasa Marine Reserve is approximately 12km^2. It took several years for these regulations to be implemented and the exclusive use of permitted fishing gears started in 1994 with varying levels of enforcement success (McClanahan and Kaunda-Arara 1996, McClanahan pers.comm.).

McClanahan and Kaunda-Arara's (1996) underwater study of fish populations showed large differences between protected and unprotected reefs. Most fish showed some recovery inside the park after fishing was banned. Surgeonfish (Acanthuridae) decreased in biomass per hectare from 1988 to 1992, but then increased rapidly to 1994 in the protected area. Triggerfish (Balistidae) increased between 1988 and 1994 by 5 times. A similar pattern was observed in butterflyfishes (Chaetodontidae) and wrasse (Labridae). Emperors (Lethrinidae) increased rapidly between 1988 and 1992, decreased slightly in 1993 and then increased again. Parrotfishes decreased initially between 1988 and 1992, then increased to above the initial level in 1993 and 1994. The largest size classes of fish in all the families studied were not present in counts on unprotected reefs. Fish biomass was lower on unprotected compared to protected reefs for all families studied, except for porcupine fish (Diodontidae). Most differences in biomass were of order of magnitude scale. For example the total estimated wet weight of fish for protected areas was 1400kg/ha but only 90kg/ha on unprotected reefs, and 200kg/ha in the Mombasa Marine Reserve where traditional fishing was permitted.

McClanahan and Kaunda-Arara (1996) compared catch data collected before the park was protected from fishing (8 months from January 1991) with data collected over the first 2 years after the protection was enforced. They found that CPUE initially increased significantly by 110% after the no-take area was created, from 20kg per person per month to 43kg per person per month (Mann Whitney U-test P < 0.05), but that total fish landed decreased significantly by 35% because of the decreased number of fishers. Total catches of 'scavenger' fish

(emperors, snappers [Lutjanidae] and grunts [Haemulidae]) and of octopus and squid decreased after protection. The CPUE of all catch species increased, but the increase was only significant for rabbitfishes (Siganidae). Rodwell (2001) summarised monitoring data for catches and biomass from inside and outside the park from 1994 to 2000 (using data from McClanahan and Kaunda-Arara 1996, McClanahan and Mangi 2000, and McClanahan unpubl. data). In Mombasa Marine National Park there was a three to four fold increase in biomass from 180kg/ha in 1987 to 610kg/hectare in 2000 (biomass peaked in 1994 at 1140kg). In the partially protected reserve the biomass remained fairly constant (180kg/ha in 1987 and 120kg/ha in 2000). At an unprotected site (Vipingo) outside the park, biomass also stayed approximately constant – 60kg/hectare in 1987 to 70kg/hectare in 2000. Fish biomass in the park was significantly higher than at the partially and unprotected sites (2 way ANOVA p < 0.001). There was no significant difference between the other two sites. Parrotfish and surgeonfish made a good recovery in the park – increasing from 9% of the biomass in 1988 to 47% in 2000. Snappers (Lutjanidae) oscillated in the park, peaking in 1993 and 1996, while at the fished site the biomass of snappers was close to zero for the duration of the study.

One very important effect of protection from fishing in East Africa is to enhance reef habitat. Carreiro-Silva and McClanahan (2001) describe how overfishing of triggerfishes, predators of sea urchins, has caused great increases in urchin densities and grazing rates in fishing grounds. This leads to high rates of erosion of reef substrate by scraping urchin mouthparts, which reduces coral cover and reef structural complexity. By protecting predators of urchins, parks can reverse such effects. Coral cover increased from 10% in the Mombasa MNP in 1987 to 47% in 1996, but decreased dramatically in 1999 in response to the coral bleaching of 1998 (Obura 1999). Hard coral cover increased again from 12% to 18% from 1999 to 2000. Hard coral cover did not show any initial increase in the reserve (partially protected), but at Vipingo (unprotected) it also increased, from 18% in 1987 to 32% in 1993 (McClanahan and Obura 1995, McClanahan et al. 2001, McClanahan unpubl. data, Rodwell 2001).

McClanahan and Kaunda-Arara (1996) also compared catches from a landing site close to the park (<2km) and one further away (approximately 8km). They found that total catch per area and catch per unit effort were much higher closer to the park than further away. Yields increased in the adjacent reserve for a short time during this initial study period, but by the end of the study catches were about the same as they had been before the park was created. McClanahan and Kaunda-Arara (1996) attribute this to the increase in fishing effort at the edges of the park which prevented the more widespread dispersal of fish from the park (protected) throughout the reserve (fished). They also point out that they did this study less than a year after effective enforcement began in the reserve, giving little time for effects to become manifest. McClanahan and Kaunda-Arara (1996) also consider briefly the economic effects of the park. They estimated income through fishing to have declined from 1991 to 1994, but total income to have remained steady because of increased income derived from park fees (and possibly tourist boats).

McClanahan and Mangi (2000) looked at spillover of fish from Mombasa Marine National Park in 1996-7 after 5-6 years of effective protection. Catch per fisher and catch per area adjacent to the park increased significantly, by more than 50%, but total catch was reduced by 30%. The protected area is mainly lagoon habitat and fringing reef. In the reserve, fishing is mainly restricted to the nearshore lagoon, with only a small amount of fishing effort on the windward reef because of the small boats used and their restricted travelling distance. The large tidal range and movement of water between the lagoon areas and the reefs meant that there was likely to be a kind of spillover effect from the outer reef too, so they designed an experimental trap fishing programme to compare spillover from the park with that from the less fished reef edge.

McClanahan and Mangi (2000) placed their experimental traps mainly in areas of seagrass, coral rubble and sand (habitat did not change significantly with distance from the park). They looked at the southern and northern borders of the park and found that with increasing distance from park boundaries, biomass of fish per trap, mean size of fish and number of fish species caught per trap declined. The decline was more rapid on the northern side than on the southern side. They used another assessment of fish density – a fish bite frequency assay on the seagrass *Thalassia hemprichii* – and found a variable but negative relationship between numbers of fish bites and distance from the reserve, implying spillover for herbivorous fish species.

Fish biomass, size and species diversity declined with increasing distance from the back-reef edge at the southern boundary of the park, but not at its northern boundary. McClanahan and Mangi (2000) conclude that spillover of exploitable fish was a significant, though variable influence on the dynamics of the fishery – part of a complex interaction with fishing gear, habitat characteristics and tidal patterns. They attribute stronger spillover effects at the southern end of the park to different fishing practices and reef morphology. The species that demonstrated strongest spillover effects were considered to be 'moderately mobile', for example rabbitfish, emperors and surgeonfish. This is what would be expected – the biggest changes would be expected in fish

species that are mobile enough to disperse but site-attached enough for stocks to increase in response to closed area protection. Fisher behaviour also supported this evidence for a spillover effect. On the southern boundary of the park, density of fish traps increased approaching the park edge (McClanahan and Mangi 2000). In interviews fishers gave the proximity of the park edge as just one of a number of attributes they considered when deciding where to set their traps. Fishers now operate a hierarchy whereby the most senior fishers fish nearest to the park boundary, confirming that they believe that catches are better near the park (Rodwell 2001).

McClanahan and Mangi's (2000) research also suggests that fish moving in from the reef edge also play a role in maintaining the nearshore fisheries. Limited travelling capacity of local fishers means that fishing effort on the reef edge is very low, so this area acts in a similar way to a no-take marine reserve as well. In fisheries like the one adjacent to the Mombasa Marine National Park, much of the catch is made up of smaller adults and juveniles, but the reproductively active adults are relatively unexploited (mainly on the reef edge) and so can continue to supply recruits to the fishery. Expansion of the fishery offshore could cause further declines in nearshore fish catches.

McClanahan and Mangi (2000) did not find that catches in the Mombasa Marine Reserve increased over time, rather that they changed rapidly in response to changes in management regimes and use of fishing gear. One important prediction about marine reserves that was confirmed, however, was that variation in fish catches decreased with the creation of the no-take marine park. From their data McClanahan and Mangi (2000) suggest that spillover effects may have only extended for a few hundred metres in the north of the park, but up to 2km at the south of the park. Data from one landing site on the southern side of the park suggests that spillover could have increased catch per fisher by up to 75%. The results indicate that in the first decade of the use of the no-take area, there were localized fisheries benefits. Broader scale effects of export of offspring of fish from the park were not assessed.

Total fish landings were lower at the end of McClanahan and Mangi's (2000) study (which ended in 1998) than they had been at the beginning of effective management, 6-7 years earlier. The authors suggest that one reason the Mombasa MNP has not had a more dramatic positive effect on overall catches is because with the protected area forming such a large proportion of fishing grounds for the villages involved, 7 years of protection may not have been enough time for stocks to build up sufficiently to compensate. It may also be that in this type of set up, a network of smaller no-take zones encompassing the same total area may have been more effective (McClanahan and Kaunda-Arara 1996 suggest that the park had a low ratio of park edge to area).

In a further study of fish landings at sites near the Mombasa Marine National Park and at other sites in the south of Kenya, McClanahan and Mangi (2001) found that landings at all sites had declined between 1994 and 1999, even though total fishing effort had apparently remained constant. At Kenyatta beach, the main landing site for people fishing in the reserve adjacent to the MMNP, after 3-4 years of effective reserve management, aggregate catches were declining at an annual rate of 6kg per day and catch per fisher was decreasing by 320g per day. Much of the decline was in rabbitfish. Of the five landing sites assessed, Kenyatta did not have the highest catch per fisher – this was found at Galu landing site where there had been a long-term ban on beach seine nets. Nonetheless, the mean catch also dropped in the last couple of study years at Galu as a result of increasing effort. However, they did find that Kenyatta had the highest catch per area of all the sites studied and its rate of decline was also lower compared to those from the other sites (250g/day compared with 310-400g per day at the other sites). Theoretical studies suggest that the fraction of fishing grounds protected needs to be larger the more intensive the adjacent fishery (e.g. Pezzey et al. 1999). It is possible that the continuing decline of fisheries near the Mombasa Marine National Park is because the park is not sufficiently large to supply such an intensive fishery. It should be remembered that the fishery was overexploited at the outset of management and was unlikely to have been sustainable (Rodwell 2001). Catches in 2000 are therefore probably higher than they would have been without the park.

Catch data (from McClanahan and Kaunda-Arara 1996, McClanahan and Mangi 2000, and McClanahan unpubl. data in Rodwell 2001) showed that catches at Kenyatta landing site decreased from 60 tonnes per year in 1991 (but based on a high estimate of number of active fishers) to 15 tonnes per year in 1994 and have oscillated between 20 and 30 tonnes/year, with 29 tonnes in 2000. Catch per unit area increased from 8 tonnes/km^2/year in 1991 to 15 tonnes/km^2/year in 1992, then fell to 6 tonnes/km^2/year in 1994, then remained stable between 1996 and 2000 at around 7 tonnes/km^2/year. Catch per fisher per day increased in the first year (1991 to 1992) from 2kg/day to 5kg/day in response to the sudden decrease in fisher number, then fell again to 2kg/day in 1994. Since 1995 the catch per fisher per day has declined as the number of fishers increased again at the Kenyatta landing site.

Rodwell et al. (2002) used new fisher interviews and catch data combined with data from other studies (Glaesel 1997, Ngugi 1998) in their model to assess the effects of the MMNP on the reef fishery. Their model predicts that equilibrium catch and fish biomass levels would be reached 10-15 years after full protection began, but that initial declines might be observed depending on the exploitation rate in the fishing grounds. Rodwell et al. (2002) estimates the exploitation rate in the Mombasa fishing grounds of between 48-80%. In a 60% exploitation scenario they predicted that catches would fall for two years after protection, then start to improve but not reach the initial level of total catch. This is what has been reported to have happened so far in the initial 6-7 years of the implementation of the MMNP.

Kisite Marine National Park

The Kisite Marine National Park (KMNP) is a no-take area that is located further south in Kenya, about 30km from the Tanzanian border, offshore from the village of Shimoni. Next to the park is the Mpunguti Marine Reserve where traditional fishing is allowed, and the total area of the park and the reserve is approximately 23km^2 (Watson and Ormond 1994). The whole area, including the reserve was designated as a no-take park in 1978 but the reserve area was reopened to traditional fishing in 1988. Beach seine nets, spearfishing, shell and coral collecting are forbidden in the reserve (Malleret-King 2000).

Watson et al. (1997) studied the effects of protected areas on fish populations and fishery catches at three marine reserves in southern Kenya, including Kisite Marine National Park, between 1992 and 1995, after 3-6 years of effective protection. They found that populations of commercial fish families were higher in the unfished KMNP compared to the adjacent fished Mpunguti reserve. They also found that abundance of fish increased over time between 1988, when Samoilys (1988) carried out an initial survey, and 1992 to 1994 when Watson et al. (1997) carried out their surveys. Catches from the Mpunguti reserve declined in the 1980s, but since 1989 when enforcement began to improve in the park, catches made a substantial recovery. The banning of seine nets probably also contributed to this recovery (T. McClanahan pers. comm.). Watson et al. (1997) also looked at the proportion of herbivores and predators in catches and concluded that the unfished park was acting as a refuge for large predators and that these populations were replenishing the adjacent fishery. They found that predatory fish dominated the artisanal catch from Mpunguti, and that there was a lower proportion of predatory fish (emperors, snappers and groupers) in fishers' catches than they found in experimental catches made inside the Park. Length frequencies of fish from the park and reserve show smaller average sizes in the fished reserve.

In a study of the movement of a key fishery species, the emperor *Lethrinus borbonicus*, Watson (1996) found no evidence for large-scale movements of this species between the Kisite Marine National Park and the Mpunguti reserve. Watson found that the movement of this species appeared to be polymorphic, with most individuals having ranges of about 60m but some moving several hundred metres. However, only 5.2% of tagged fish were resighted despite quite intensive effort and some may have made more extensive movements. From this study Watson (1996) concluded that the main fishery benefits of the park were likely to be through larval export rather than adult migrants, at least for this emperor species. One key point about the fished and unfished reef areas in this study was that the reef was not continuous between the two. They were separated by about 2km of sandy substrate which would have limited movement of some fish species between the two areas. Watson (1996) suggests that to optimise benefits to reefs and fishers several small closed areas would be better than one large one and that the closed and fished areas should be linked by continuous reef (as is the case for most of the reserves in the SMMA in St Lucia and to a lesser extent at Nabq in Egypt; see case studies).

Watson (1996) investigated the potential for larval dispersal from Kisite using a drift bottle study and found that pelagic larvae could potentially be carried several hundred kilometres, assuming passive transport. Watson (2002) also points out the importance of Kisite's position for supplying the reefs further north along the coast as the prevailing currents run north, thus widening the potential benefits to many other protected and fished reefs in the south of Kenya, and perhaps beyond.

Watson et al. (1997) also looked at fish numbers underwater in two other Marine parks in the south of Kenya (Malindi MNP and Watamu MNP) and found that fish abundance for the most important commercial species had fallen since Samoilys' (1988) surveys of these sites in 1988. They also found that fish abundances in these marine parks were not significantly different from those in areas open to fishing. They attribute this lack of difference to poor enforcement of these parks in the past, compared to the relatively effective enforcement at Kisite. They suggest that this was due in part to the need for fishers to travel through the Malindi and Watamu parks to get to fishing sites, making it easier for fishers to fish opportunistically in the no-take areas. This was not necessary at the offshore Kisite park, and also the enforcement office was in close proximity on Mpunguti island, overlooking the closed area at Kisite.

Emerton and Tessema (2001) looked at economic aspects of the KMNP and the Mpunguti reserve. They present data on revenue generated by the park which is one of the most profitable in Kenya. In 1998 just under 30,000 visitors paid an entrance fee of US$5, generating over US$131,000. This money goes to the general Kenya Wildlife Service funds and is redistributed amongst all parks and reserves they operate. Watson (2002) points out that this means that communities associated with profitable reserves and parks like Kisite do not see much of the revenue generated in their area as much of it feeds back into less profitable areas. In 1998 US$19,000 was spent running KMNP/MMNR, including allocations for benefit sharing projects in the wider community. Some funds were allocated to community projects through KWS Wildlife for Development Fund, funding school improvements, supporting the boat operators association, donating fishing equipment and repairing a fish depot (Emerton and Tessema 2001). Emerton and Tessema (2001) estimate that the budget required to run the KMNP/MMNR effectively would be US$135,000 per year.

From an estimate of maximum sustainable commercial fish yield, Emerton and Tessema (2001) estimate the loss of potential catch from the KMNP to be 147 tonnes per year with a value of US$172,000 at 1999 prices. However, there are many problems with this estimate not addressed by the authors (M. Watson pers. comm.) and the figure is unreliable. Additionally, Emerton and Tessema (2001) were not able to measure or include potential regional fisheries benefits from export of larvae from the reserve. Such effects could be very important considering the intensively overexploited condition of many of Kenya's fisheries (Rodwell et al. 2002). Furthermore, Kisite clearly generates countrywide benefits by raising revenue for Kenya's parks system.

Emmerton and Tessema (2001) say that most fishers feel excluded from tourism benefits and claim that attempts by local people to make money from tourism through boat services, souvenir sales and tours have not been particularly successful. However, hundreds of local people are employed in tourist operations, in restaurants, as boat captains, crew, dive guides and other work. There is a scheme whereby each park visitor to the restaurant at nearby Wasini makes a direct contribution to a fund for Wasini village. All the fish for this restaurant (up to 100 people a day) is bought locally (M. Watson pers. comm.).

Malleret-King (2000) took a different approach to evaluating the effects of the Kisite Marine National Park on the fisheries and fishing community. She looked at its effects on the community in terms of how their food security had changed in response to the management of the reefs. She found that fishing households were the least food secure in the communities, but that fishers who fished nearer the park had better food security than those who fished further away. The most food secure families were those who obtained their income from tourist sources from the park. Overall her results showed that the benefits of the KMNP were not distributed equally amongst the communities that were bearing the costs of the park. Households nearest the KMNP were more likely to derive some of their income from tourism. Fishery dependent households were separated from other households who derived some income from tourism, and the relationship between the park and the fishery was investigated in more detail. Short term and longer term coping capacities (the capacity for the household to cope with shortages of food - regarded as amongst the best indicators of food security) were found to be higher for households using fishing grounds closest to the park, suggesting that the communities fishing closest to the park may have benefited from spillover effects. Access to fishing grounds adjacent to the park varied between communities, and this meant that the potential for fishers who had lost fishing grounds in the initial designation of the park to benefit from spillover effects varied greatly.

The distribution of benefits is an important factor to bear in mind when considering other results showing increased catches and other benefits of marine reserves – are the people who initially made the sacrifice benefiting in proportion to their initial losses? This is another angle from which to consider the compensation question, and another complicating factor. Malleret-King (2000) suggests that approaches need to be found to compensate the appropriate communities or to share the benefits of marine reserves to enable communities to support and participate in their protection. The key to survival in many rural coastal communities is diversifying incomes. One approach to improving the situation for those benefiting less from the marine park might be improved farming methods and land tenure system (Malleret-King 2000). Increasing opportunities for agricultural diversification was one of the approaches successfully used at the Nosy Atafana Marine Park in Madagascar (Grandcourt et al. 2001, see case study).

In discussions with fishers from five different villages using fishing grounds adjacent to the Kisite park, Malleret-King (2000) found that most of the fishers had negative views of the park. Only one group identified a positive effect on the fishery, saying that they caught bigger fish near the boundaries of the park. The fishers identified the associated aid programme that accompanied the introduction of the park, including the provision of boats with engines, as the main positive aspect of the park.

Malleret-King's (2000) study is more detailed than Emmerton and Tessema's (2001) and draws on an extensive base of work with local communities. Contrary to their findings, Malleret-King confirms what is often suggested about the use of no-take areas, that one of the major benefits to a local community will be income from increased tourism. However, tourism is often seasonal. In Kisite the low tourist season corresponded with the bad fishing season, thus exacerbating seasonal fluctuations of income and opportunity (there is a similar pattern in the Soufrière Marine Management Area in St Lucia, with the more lucrative offshore fishing season corresponding with the best tourist season). Malleret-King (2000) points out the importance for tourism development in association with a marine reserve to be approached with sensitivity to possible environmental and social impacts of that development. Caution is needed in the extent to which communities become reliant on what can be a fickle industry.

Key points

- Fish biomass has increased in the Mombasa and Kisite Marine National Parks since they have been protected, and they have higher fish biomass than comparable unprotected and partially protected areas nearby.
- At Mombasa CPUE increased but overall catch decreased due to a reduction in the number of fishers.
- At Mombasa there is spillover of fish from the park to adjacent fishing grounds, with an estimated range of influence of a few hundred metres to 2km from the park.
- The strongest spillover effects were observed for moderately mobile fish families, such as rabbitfish, emperors and surgeonfish.
- Variability in fish catches in adjacent fishing grounds reduced in response to the Mombasa Marine National Park.
- Marine parks at Mombasa and Kisite have promoted tourism, and have slowed the decline of adjacent fisheries, compared to rapidly declining unprotected fisheries.
- Fishers in the vicinity of Kisite MNP have higher household food security if they fish near to the park, while fishers further away did not do so well, and were also less likely to derive additional income from tourism.
- Enforcement can be made effective by logistical details such as the proximity of enforcement offices.
- Benefits from marine reserves in Kenya are not shared equally amongst users, and those who lost out initially do not necessarily benefit most from either increased fish CPUE or from increased tourism opportunities.
- Kisite generates significant income from tourism but little of this revenue returns to the area. The park needs more money for day to day running and more money should come back to the fishing community.

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12. The Soufrière Marine Management Area, St Lucia, West Indies

Soufrière is a small town on the south west coast of St Lucia for which fishing and tourism are important sources of income. The coral reefs that fringe the coast there form the basis for much of this income. Most of Soufrière's tourism is concentrated around snorkeling, diving and yachting, and the local fishery is divided between the nearshore reef fishery and an offshore fishery for pelagic species like tuna. Since the 1980s there has been conflict over the limited nearshore resources, between fishers and tourist use, and concern over declining reef fish catches and reef health.

In 1986 the Government of St Lucia declared many of the major reefs in St Lucia as marine reserves, with legislation to protect them from fishing and other sources of impact. However, the protected areas were not officially defined, and there were no funds available for marking boundaries or enforcement, and so the legislation was ineffective (White 1994, George 1996). In 1987 the Government put forward a proposal for the development of a National Park in the Soufrière area, to include both marine and terrestrial habitats, focussing particularly on the coral reefs. Various attempts to establish reserve and fishing priority areas were made by the Department of Fisheries. Local groups began consultation on the areas to be protected and they put a monitoring programme in place but there was no active management, with the exception of a small no-take reserve area established in front of one hotel (the Anse Chastanet reserve – see Roberts and Hawkins 2000).

In 1992 the various parties interested in management of Soufrière's marine resources renewed their efforts to establish a marine and coastal resource management area. They formed a committee with representatives of the Soufrière Regional Development Foundation, Department of Fisheries and Caribbean Natural Resources Institute. This committee identified stakeholders to be involved in the new consultation process and co-ordinated negotiations (George 1996). The committee decided that the best approach would be to start with a clean slate, putting all existing marine reserve and fishing priority areas under review. Resource users and other stakeholders produced maps identifying current uses, conflicts and condition of coastal resources around Soufrière. Various issues were taken into account, including yachting and diving, fishing, sources of conflict, local recreational use and issues of pollution and reef degradation. After extensive community consultation the proposal for the Soufrière Marine Management Area was presented to the Government for approval in 1994, and accepted. The stakeholders, together with local and regional experts put together a management plan. USAID and the French Mission for Co-operation provided support for the initial management of the area, funding demarcation of the reserves, moorings for yachts, a well equipped office, patrol boat and initial salaries for a manager and wardens (George 1996).

The Soufrière Marine Management Area (SMMA) was officially launched in 1995, with effective management beginning in July 1995. The management area covers 11km of coast and extends 100m from shore, or to 70m depth, whichever is greater, and is separated into a series of zones including:

1. Marine reserves – all extractive uses are forbidden, but diving is permitted.
2. Fishing priority areas – where diving and other uses are permitted but fishing takes precedence.
3. Yacht mooring areas – areas where there are mooring buoys provided for yachts. Fishing and diving are permitted.
4. Multiple use areas – all uses are permitted in these areas, with the exception of activities forbidden throughout the SMMA such as jetskiing, and coral extraction.

One of the main objectives for establishing the SMMA, incorporating no-take marine reserves and fishing priority areas, was to manage the local reef fishery and prevent further decline in reef fish catches and the health of the coral reef ecosystem. The Soufrière Fishermen's Co-operative (SFC) took an important role in the SMMA consultation process and approved the location of the no-take areas and the fishing priority areas. One of the problems that reef fishers later identified with the consultation process was that the active members of the SFC were mainly seine net fishers who fish off sandy beaches targeting small pelagic species such as jacks (Carangidae) and halfbeaks (Hemiramphidae). The fishing priority areas they selected were mainly beach areas, and the no-take areas encompassed many good reef fishing areas. The reef fishers thus felt that they had not been represented fairly and had lost most of their good fishing grounds, whereas the seine fishers had retained the use of all their best sites.

Protection of reef fish stocks has been a great success. Annual monitoring of reef fish biomass since prior to the beginning of management has shown a four fold increase in commercial fish biomass inside the marine reserves and a three fold increase in fishing grounds (Figure 1, Roberts et al. 2001, Gell et al. in prep. a).

Goodridge et al. (1997) collected baseline data on the status of the Soufrière reef fishery at the time of the implementation of the SMMA and during the early months of management. Roberts et al. (2001) compared this information with data collected in 2000/1, after five years of effective management. They found significant increases in total catches and catch per unit effort of reef fishers using the two main fishing gears – large traps set overnight and small traps baited with fish and set for 1-2 hours (Fig 2). For big pots, mean catch per trip, catch per pot per trip and catch per hour at sea all increased significantly, whilst there was no significant change in the mean catch per day soaked for traps. For small traps mean catch per trip and catch per pot per trip increased significantly, whilst there was no significant change in mean catch per hour fished. Catch per unit effort increased by 46% for large traps and 90% for small traps. Since overall fishing effort was similar between 1995/6 and 2000/1, Roberts et al. (2001) concluded that reserves had increased total reef fish landings above levels at the time of implementation of the reserves.

In terms of catch composition, in the large pots the proportion of surgeonfish (Acanthuridae), snappers (Lutjanidae) and grunts (Haemulidae) increased. In small pots, which generally targeted smaller fish, there was an increase in the proportion of groupers (Serranidae), squirrelfish (Holocentridae) and parrotfish (Scaridae) (Gell et al. in prep. b). In 2000/1 fishers caught over 100 species of fish from Soufrière's reefs and snappers, parrotfish, squirrelfish, groupers and moray eels (Muraenidae) dominated catches. In the underwater surveys of fish, snappers and parrotfish showed the clearest reserve effects, with rapid increases within the reserves, followed by increases in fishing grounds. The diverse range of exploited fish species showed a wide variety of responses to the closed areas, inside and outside the reserves (Gell et al. in prep. b). Some responded rapidly to protection, others increased after a delay, and a few have not responded at all.
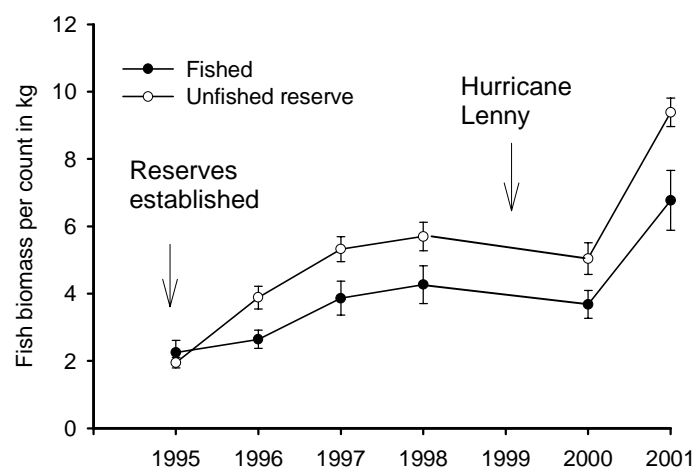

Figure 1: The mean biomass of commercially important fish per count in kilograms in fished and unfished areas of the SMMA from 1995 to 2001. Error bars show +/- SE.

One of the reasons for the success of the SMMA in terms of increasing reef fish catches is the network design, with four main areas of no-take reserve, interspersed with fishing priority and multiple-use areas. Most of the main fishing sites are reef areas where the reef is contiguous across no-take and fishing priority areas, providing continuous habitat across the protected-unprotected border. Fishers often place fish pots or fish with hook and line on the boundaries of the marine reserves, confirming the belief of many fishers that there are "better fish inside the marine reserves".

The reef fishers found the first couple of years of management difficult. They were unable to use some of their best fishing grounds and some had to travel further to reach fishing sites, which could be arduous considering that many lacked outboard engines, relying on oars, and currents can be strong. In 1997, after two years of

management, there were growing problems of illegal fishing by reef fishers, and intense political pressure from them to re-open marine reserves. As a compromise, one reserve area was reopened to fishing and compensation was paid to 16 of the fishers who were most reliant on the reef fishery. In general they included the oldest fishers, spanning an age range of around 50 to over 70. Compensation was significant, coming to approximately US$150 per fisher per month for 12 months, and in interviews a few fishers claimed that they were able to give up fishing while they were paid this compensation. Compensation greatly improved compliance with reserves at a critical time (K. Wulf, SMMA manager, pers. comm.), but at the end of the year of compensation the loss of that income was difficult for some fishers to accept. In economic terms, directing compensation to older fishers is sensible. This is because their 'discount rates', the degree to which fish in the hand are worth more than fish tomorrow, are higher than for younger fishers. Younger fishers have more time to reap the rewards of early sacrifices made in protecting reserves. Furthermore, younger fishers were more able to maintain yields in the early years of reserve creation by increasing their time spent fishing (Gell et al. in prep a).

In a study of the Soufrière reef fishery in 2000/1, after 5 years of management, compliance was found to be generally good, particularly amongst the majority of regular reef fishers (Roberts et al. 2001). Illegal fishing did occur though. For example, trap fishing effort in reserves constituted 19.9% of effort in legal fishing areas. However, illegal fishing was particularly a problem amongst spearfishers and people line fishing from shore. A small number of fishers reportedly fished illegally at night as well. Enforcement is the responsibility of rangers who patrol the management area twice a day. They collect mooring fees from yachts and warn fishers who have 'strayed' into reserves. Strength of enforcement varies with the rangers on duty – but persistent offenders have had fishing gear confiscated, and rangers have removed fish pots and gill nets found set illegally in the reserves. However, the rangers are less interested in illegal fishing by line fishers on shore who are often difficult to see from the water, and these fishers are rarely even warned.
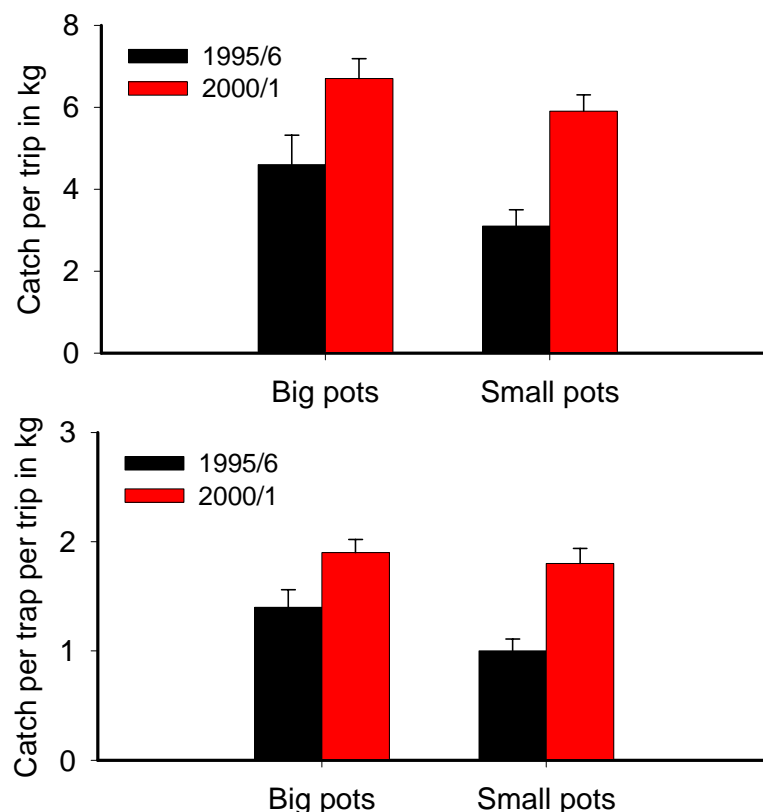
Figure 2: Mean catch per fishing trip and mean catch per trap per trip (bottom) for big pots and small pots fished in the fishing areas of the Soufrière Marine Management Area. Error bars show +/- SE.

There is quite a strong dichotomy in the Soufrière fisheries. Many younger men now fish off-shore using modern fibre-glass boats with large engines, often bought with loans subsidised by the Government. Many combine

offshore fishing with tourism, such as offering water taxi services, and a few are also involved in the 'informal economy' of the region. The reef fishers most affected by no-take areas are mainly older men (>50 years) with limited resources in terms of boats and gear. About half of them use traditional designs of boat such as wooden canoes or small rowing boats, and do not have engines. These more traditional fishers are also limited in the other incomes they can make. While younger fishers can adapt easily to fishing offshore and serving the tourist industry, this is much more difficult for many older fishers who feel that their only vocation is reef fishing and they are too old to change.

The costs to the reef fishers have included the loss of 35% of potential fishing grounds, including some of the sites closest to the main town where many of them live. They have therefore had to either invest more money in transport costs, such as fuel, or spend more time rowing to sites. In recent years, some fishers have started to fish in deeper waters (>70m) not regarded as part of the protected area. For line fishers this requires long lines and hard work hauling up fish, but it is particularly hard work for trap fishers who all haul traps by hand, and they have to invest in more rope. Fishers claimed that they caught less fish and things were harder for them in the initial couple of years of management. Although compensation obviously helped, it was only for one of 2-3 years of hardship. Catch statistics collected by the Department of Fisheries confirmed the decreases in catches talked of by fishers in the initial years, followed by significant increase in catches after 4-6 years of management (Hubert 2001, Gell et al. in prep. a.). The direct benefits that reef fishers have derived from the SMMA include increased catches and CPUE after five years of management, support and logistical help from the SMMA with finding lost fishing gear, and a local representative to go to for support in conflicts with other user groups. For example, the SMMA authority have supported fishers in their appeals to dive operators to prevent divers tampering with fish pots, including a poster campaign urging tourists to respect the local fishers and their fishing gear. In another example, fishers reported a tourist boat dumping garbage in the bay at night and the SMMA authority dealt with this. Acting on their own, fishers would have been less likely to see such activities dealt with.

Indirect benefits to reef fishers have come through improved opportunities for income from the tourist industry. Some reef fishers have become involved in the tourist industry themselves, but others feel excluded because of the need for capital to buy the appropriate boats, or socially excluded because they are older and uncomfortable speaking English and interacting with tourists. However, in some cases the household income of these fishers has improved through other family members taking jobs in tourism. One problem in Soufrière has been fluctuations in tourist numbers, and employment in most areas of the tourism industry is seasonal and not particularly secure. When tourism jobs are lost many people turn to casual reef fishing and illegal fishing increases. Tourism does have another indirect benefit to fisheries. The SMMA is funded through user fees levied on scuba divers, snorkelers and yachts. Hence tourists pay for the management that has brought fishers improved catches.

An important development as a result of the SMMA was the establishment of the Soufrière Water Taxi Association. This organisation has formalised the involvement of boat owners and others in the water-based tourist industry and has introduced systems of operating to protect both their members and the people using their services. They ensure that boats used for taking tourists have appropriate safety standards, and members benefit from a central office for bookings and pooled resources for things like advertising, producing leaflets and entries in tourist guides. The Association also provides strong representation on local committees, such as the SMMA management committee and local tourist and development committees.

The Soufrière Fishermen's Co-operative also has a strong link with the SMMA, but according to some of the reef fishers, still does not represent them adequately. Fishers have to pay to join the FSC and the main advantage of joining is a tax rebate on boat fuel. Therefore, older fishers who row their boats and don't have an engine aren't members. The more active members of the co-op are seine net fishers or offshore fishers.

In interviews with Soufrière fishers, differences were evident between generations. Older fishers tended to be much more reliant on the reef fishery and had fewer income alternatives, and in general showed the least support for marine reserves. Younger fishers were less reliant on the reef fishery, benefited more from alternative sources of income in tourism and were more supportive of marine reserves in interviews. However, compliance was often better amongst the older regular fishers, and most illegal fishing witnessed was by younger casual fishers, often deep-sea fishers on their way home from a bad fishing trip or tourist workers in the low season.

One issue raised by reef fishers was that even in the light of improved catches, they still saw the SMMA as primarily for the benefit of tourists and foreign-owned tourist operators. They felt it was unfair that there were areas that were closed to fishing, but none that were closed to diving. There is actually one area that is closed to diving but it is right at the south of the SMMA, far from most of the fishing activity by fishers from the main

community. Many fishers felt that either fishing priority areas or reserves should also be closed to divers, or that the activities of divers should be more closely monitored to prevent damage to fishing gear. Despite some of these continuing problems, the zoning of the SMMA has been successful in separating conflicting activities and making it clearer to all users who has precedence in which areas.

There have been a number of set backs in the development and success of the SMMA. In 1999 waves from Hurricane Lenny hit the area. Soufrière Bay was severely affected, destroying waterfront buildings, including many fishers' homes, and also destroying fishing boats and gear. Hurricane waves also seriously damaged some areas of reef, and Roberts et al. (2001) reported a slight downturn in fish stocks after the storm (Figure 1). There was an increase in illegal fishing after the Hurricane, as a result of hardship amongst the fishing community, but at the same time some regular fishers were unable to fish because their boats were lost or damaged.

The SMMA has provided a focus for community education about environmental and marine issues, and interviews with fishers from Soufrière revealed familiarity with many key fisheries management and conservation concepts. The SMMA provides a number of school and community education programmes, and having rangers and office staff recruited directly from the fishing community means that much of the conservation and management messages are reinforced informally too. Many fishers mentioned the importance of taking care of coral reef habitats, not catching juvenile fish and leaving enough fish to produce the next generation. One of the most interesting tactics that was used on the enforcement and education side was to recruit as a ranger one of the fishers who was most vocally and actively opposed to the marine reserves. He has now become one of the most effective rangers, and is now as vocal and active in his support of the reserves as he was previously in his opposition.

In interviews in 2000/1, many fishers felt that the SMMA authority could give them more information and there did seem to be a need for more every day interaction between fishers and the SMMA, for example with a community liaison or education officer. It was also evident that awareness and understanding of the role of the marine reserves in managing the reef fishery was much lower in fishers from outside Soufrière town, where most of the education programme had been focussed. There was a particular problem with illegal fishing by seine net fishers from a village further north targeting halfbeaks in the no-take areas, and also with fishers from small rural communities fishing with lines on the shore in the reserves. This generated resentment amongst the local fishers and some suggested that it encouraged Soufrière fishers to fish illegally as well.

The reopening of one area of reserve in 1997, initially only to a particular group of fishers, had repercussions that were still evident in 2000/1. The opening caused confusion, with some fishers believing that all reserves would be opened at some point, in a rotation scheme, which was never the intention. Other fishers believed that the opening of one area meant that another nearby reserve area was also open to fishing. One fisher claimed that he had never realised that the area had been formally opened and that he thought people were fishing there illegally. The fishers who were permitted to fish in the reopened area claimed that it was not a particularly good area anyway as strong currents limited access. This kind of confusion emphasises the need for continuous contact and reinforcement of the rules of marine reserves and the reasoning behind them.

The SMMA has succeeded on two levels – firstly it has sustained and enhanced the local reef fishery which was previously declining, increasing the catches of local fishers. Secondly it has promoted conservation of coral reefs and sustainable development of tourism in the area. There have been some problems with the fisheries objectives, in that full representation of the fishing community was not achieved in the initial consultation process, but this has improved over the years. Most reef fishers now support the SMMA. The wider conservation and development aspects have been achieved because the SMMA was a community driven initiative that involved a wide cross-section of stakeholders and interest groups in consultations. Compensation played an important role in ensuring the welfare of displaced fishers for one year at a critical time when catches initially fell. It also improved future support of the SMMA, increasing compliance with regulations.

Key points

- The SMMA is an example of a failed "paper park" transformed into active and successful management.
- Biomass of commercially important reef fish increased by 3 times in fishing grounds and 4 times in no-take marine reserve zones.
- The network of small marine reserves has increased catch and CPUE in the reef fishery. This design – small reserves interspersed between fishing grounds – probably played an important role in its success.
- Compensation for one year played an important role in sustaining the marine reserves when catches were poor and morale was low.
- The zoning plan has been successful in reducing conflict between fishers, divers and yachts.

- The SMMA would have been improved if the initial consultation had involved a wider cross-section of the fishing community. Fishers are a diverse group with a variety of needs and agendas.
- Rangers recruited from the local fishing community can be very effective.
- After five years of ups and downs, wider support from reef fishers was gained, and is growing.

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13. Gulf of Castellammare trawl ban, Sicily

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In 1990 the Sicilian Regional Government implemented a year-round trawl ban over an area of 200km^2 in the 400km^2 Gulf of Castellammare in northwest Sicily in the Mediterranean Sea. The closure comprises 55.5% of the whole Gulf and its purpose was to rebuild fished stocks in the area and to address the conflict existing between the 15 or so commercial trawling vessels and the 150 artisanal vessels using fixed gears. There was also conflict between the artisanal fleet and the recreational fleet, and the future of artisanal fishing in the Gulf was in doubt. The fishing ban only applied to towed bottom gear. Artisanal and recreational fishing using other methods were still permitted in the area. Based on experimental trawl surveys and landings data, before the ban was implemented the resources were probably overexploited. The trawl closure is enforced by coastguards, but poaching has become more common in recent years, especially when the weather is bad and there are less likely to be police patrols (Pipitone et al. 2000b, 2001b).

To assess the effects of this closure, Pipitone et al. (2000a) conducted trawl surveys one year and three years before the ban (1987 and 1989) and 4 years after the implementation of the ban (1994). They used experimental fishing, in the form of 30 minute trawls. Catch per unit effort (CPUE) measures for 11 commercial target species were analysed from 21 spring hauls before the trawl ban (1987-89) and 30 spring hauls after the trawl ban in 1994. Hauls were made in three depths: 10-50m, 51-100m and 101-200m. Seven out of the eleven species increased significantly over the time the ban was in place. Angler fishes (*Lophius budegassa* and *L. piscatorius*) increased by nearly 11-fold from 0.29 kg per 30 minute haul to 3.15 (P<0.01), comber (*Serranus hepatus*) by 35-fold from 0.03 kg haul^{-1} to 1.13 (P<0.001), gurnard (*Lepidotrigla cavillone*) by 185-fold from 0.02 kg haul^{-1} to 2.98 (P<0.001), hake (*Merluccius merluccius*) by 4.5 fold from 0.47 kg haul^{-1} to 2.21 (P<0.001), pandora (*Pagellus erythrinus*) by 5.5 fold from 0.25 kg haul^{-1} to 1.34 (P<0.05), picarel (*Spicara flexuosa*) by 83-fold from 0.01 kg haul^{-1} to 1.00 (P<0.001), and red mullet (*Mullus barbatus*) by 33-fold from 0.28 kg haul^{-1} to 9.31. Overall catch also increased significantly by nearly 10 times from 3.8 kg per 30 minute haul in the 1980s to 31.1kg in 1994 (P<0.001).

There were no significant changes in the CPUE for annular seabream (*Diplodus annularis*), the axillary seabream (*Pagellus acarne*), the horned octopus (*Eledone cirrhosa*) and the musky octopus (*Eledone moschata*), although these species increased by between 116% and 1020%.

Experimental trammel net catches made annually from 1990 to 1998 showed a significant increase in CPUE of approximately 85% from around 1.75 kg per 500m of net per 12 hours, to around 3.25 kg over the 8 year period (D'Anna et al. 2001).

Catches of artisanal fishers, mainly using trammel nets and monofilament gillnets, were also larger than before the ban (Pipitone et al. 2000b). Evidence of fishery benefits also came from the return to the fishery of fishers who had previously left when the fishery was in decline. Pipitone et al. (2000b) found that catch rates of artisanal trammel netters inside the trawl ban area were higher than those outside the ban area (6.63kg per day compared with 5.76kg per day) but that this difference was not statistically significant. Whitmarsh et al. (in press) compared catch rates of artisanal trammel netters between three ports inside the trawl ban area and one port in the region where trawling is still permitted. At an average of 5.13 kg per 1000m of net per trip, catches from the port outside the trawl ban area were only half to two thirds levels inside the trawl ban area (which averaged 7.58 to 10.26 kg per 1000m net per trip), and fishers had to travel further to reach fishing grounds to avoid conflicts with trawlers. Whitmarsh et al. (in press) note that in recent years, trawl fishers have tended to concentrate fishing along the periphery of the trawl ban area, suggesting they are benefiting from spillover.

In surveys of fishers, Pipitone et al. (2000b) found that 80% of artisanal fishers felt that fishing was "better" or "much better" since the ban. Interestingly, 55% expected the fishing in the area to get worse in the future and only 24% believed that fishing would get better. Perhaps this is because there has been a dramatic increase in the number of people involved in recreational fishing in recent years. There are now 14 recreational vessels for every artisanal vessel, leading to conflict. Artisanal fishers also fear the reopening of the trawl ban area to trawlers.

In a follow-up study four years later in 1998, Pipitone et al. (2001a) showed that the big increase happened in the initial four years of protection, and that after a further four years of protection, CPUE in experimental fishing did continue to increase but only by a relatively small amount. The increase in total catches from 1990 to 1994 was 711% and from 1994 to 1998 was 11%. Analyses of species changes from 1994 to 1998 showed that four

species decreased in the second four years (annular seabream, pandora, picarel [*S. flexuosa*] and red mullet) but by much smaller percentages than they increased in the first four years.

An additional management measure introduced with the trawl ban was a vessel tie-up of 45 days per year, imposed by law. This was initially accompanied by a compensation payment, paid from 1987 to 1997 for these 45 non-fishing days, known as a Biological Rest Payment. On average the amount received was equivalent to nearly one third of the total sales revenue (Pipitone et al. 2000b). Although the non-fishing days are still observed, payments are no longer made as they conflict with European Union law.

Key points

- Experimental fishing showed dramatic increases in total CPUE inside the trawling exclusion zone and for many of the important commercial fish species after four years of protection. After 8 years of protection experimental fishing CPUEs were still much higher than the pre-exclusion levels but had only increased by a further 11%.
- Artisanal fish catches were found to be higher inside the exclusion area but not significantly so, while experimental trammel net catches more than doubled.
- Trammel net catches were 1.5 to 2 times greater for three ports inside the trawl ban area compared to a port outside.
- Most artisanal fishers felt their catches had improved since the trawl ban.

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14. Tabarca Marine Reserve, Spain

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Tabarca Island is a few kilometres off the Mediterranean coast of Spain, close to Alicante. It is small (0.7 km^2) with a permanent population of just 50 people, but receives between two and five thousand visitors per day in summer. The main sources of income for residents are artisanal fishing and tourism. Habitats around the island are dominated by seagrass beds of *Posidonia oceanica* to 30m depth (Ramos et al. 1992). The 1400ha Tabarca Marine Reserve was established in 1986 and is divided into three use zones: a 100ha core area that is no-take, a buffer zone where some traditional fishing gears are permitted and a recreational zone where diving and other recreational activities are permitted, and some forms of fishing. Spear-fishing, boating and waterskiing are not allowed anywhere in the reserve. The reserve is marked using buoys and enforced by four rangers with two patrol boats. It is managed by a committee with representatives of the national and local government, the local council, fishers, researchers, the local scuba-diving organisation, environmental organisations and community groups (Sánchez Lizaso et al. 2001).

Eighty percent of the seabed within the marine reserve is covered in seagrass. To prevent damage to seagrass beds by illegal otter-trawl fishing an anti-trawling artificial reef has been installed in the marine reserve and a number of old boats sunk. These artificial reefs appear to have been successful in reducing illegal trawling. At another site in the area, El Campello, an anti-trawling reef on seagrass has led to increased catches of the red mullet, one of the main target species for the recreational fishery (Martínez-Hernandez 1997). Two mooring areas were also designated to reduce the impact of visitors anchoring in seagrass areas (Sánchez Lizaso et al. 2001).

Underwater surveys have shown that abundance and biomass of fish are higher in the reserve than in unprotected areas and most target species such as *Epinephelus* spp. (groupers), *Sciaena umbra* (brown meagre – Sciaenidae), *Diplodus* spp. and *Dentex dentex* (seabreams – Sparidae) have increased in abundance and biomass in the reserve since 1988 when they were first surveyed (Sánchez Lizaso et al. 2001). Exploited invertebrate populations have also recovered. At the outset of management the pen shell *Pinna nobilis* had virtually disappeared from the area, as a result of collection by divers as a curio. Now, densities are 12 times higher in the reserve than in nearby areas. Populations of octopus, squid and lobster (*Palinurus elephas*) have also increased (Ramos et al. 1992). The researchers saw improvements in the deeper seagrass habitat in the marine reserve, with a significant increase in seagrass cover in deeper waters between 1988 and 1992, but no change in shallow seagrass cover (Sánchez Lizaso et al. 2001).

Ramos et al. (1992) found that catches of a number of key species increased in fishing areas adjacent to the reserve. The catch of grouper increased by 50%, gilt-headed seabream (*Sparus auratus*) by 60% and bream (*Dentex dentex*) by 85% after 6 years of protection. Mas and Barcala (1997) confirmed these trends, finding that while total catches from the Tabarca area remained constant after the creation of the reserve, CPUE was higher near to the reserve than in other fishing grounds in the area. They found that catches of the bream *Dentex dentex* had increased by more than three fold after 9 years of protection (Figure 1).

The socio-economic status of the local fishing fleet is good compared to other fleets in the area. The Tabarca fleet is now the most modern in the region and is the only fleet in which numbers of fishers are not decreasing (Sánchez Lizaso and Giner 2001). In response to the improved viability of fishing with inshore trap nets since the establishment of the reserves, local fishers have set up a co-operative to fish with this gear, and after initial scepticism, their support for the area is improving.

Diving in the reserve is strictly controlled, and the number of divers permitted to dive the reserve per day has been reduced in response to pressure from fishers. Other forms of tourism have been increasing over recent years and many tourists come to visit the beach and village, rather than for water-based activities. A number of general benefits have accompanied development of the reserve including improvements in transport to and from the island, garbage disposal, electricity supply and other infrastructure (Sánchez Lizaso et al. 2001).

At another marine reserve around small offshore Spanish islands, the Columbretes Islands Marine Reserve, Goñi et al. (2001a,b) found significantly higher catch rates of red lobster (*Palinurus elephas*) from experimental fishing inside the Columbretes Marine Reserve, compared to two fished sites outside the reserve. The catch rate inside the reserve was between 15.1 and 29 lobsters per 40 traps, whereas outside the reserve the catch rate was 0-2.5 lobsters per 40 traps (Goñi et al. 2001a). The reserve covers 14 km^2 and lies 50km off the coast of Spain and has been protected since 1990. No quantitative data are available on the direct effects of the reserve on local fisheries but a fishery has developed around its boundary, suggesting that fishers are seeing spillover from the

reserve in the form of increased catches (Goñi et al. 2001b). Goñi et al. (2001a) reported much variation in abundance of lobster, and noted the possible seasonal movement of the lobster to deeper water. Tagging studies of lobster in the reserve suggest that it is contributing to fishery yields through the seasonal movement of adults to adjacent unprotected areas (Goñi et al. 2001a,b).
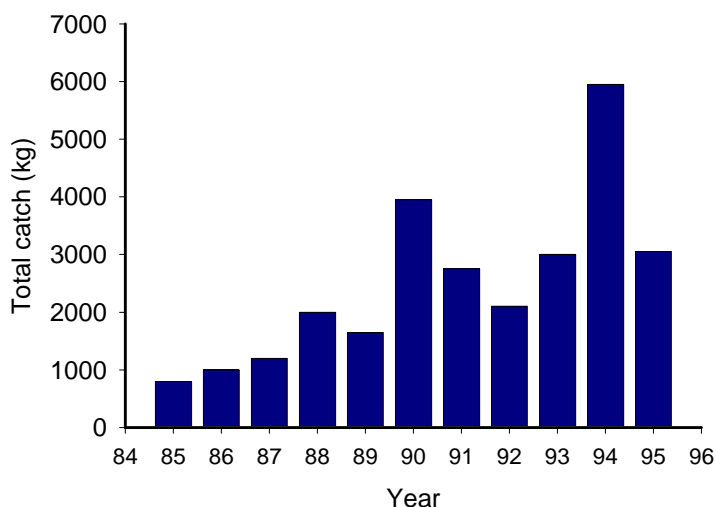
Figure 1: Total catches of the bream *Dentex dentex* around Tabarca Marine Reserve from 1985 to 1995. Protection began in 1986. Data from Ramos et al. (1992) and Mas and Barcala (1997).

Further north along the Spanish Mediterranean coast, the Medes Island Marine Reserve gives another demonstration of the effects of protection on Mediterranean fish populations. García-Rubies and Zabala (1990) studied the effect of six years of protection from all forms of fishing on fish communities and found that exploited species had a very different size structure inside the reserve compared with outside, with much larger modal size classes. Some heavily exploited spear fish species were only found in the reserve, e.g. *Epinephelus marginatus* and *Sciaena umbra*. At Medes Island there has been a dramatic increase in non-extractive uses such as diving and glass bottom boat tours (Badalamenti et al. 2000).

Key points

- Tabarca and Columbretes marine protected areas show that reserves can provide effective protection to seagrass habitats and species.
- A package of management measures at Tabarca, including a no-take zone, has apparently benefited adjacent fisheries, with increased catches and improved economic well-being for local fishers.

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15. Merritt Island National Wildlife Refuge, USA

The Merritt Island National Wildlife Refuge at Cape Canaveral, Florida, USA, contains two areas that have been closed to fishing since 1962: the Banana Creek Reserve and the North Banana River Reserve. Unlike most of the areas that we discuss in this report, they were not protected for conservation or for fisheries management purposes. The two estuarine areas that make up the refuge are closed to public access for the security of the nearby Kennedy Space Center, and have a total area of 40km^2 (16km^2 and 24km^2). They constitute approximately 22% of the total estuarine area of the Refuge which covers is 180km^2. In addition, 60km^2 of the South Banana River was closed to motorized vessels in 1990, reducing fishing pressure in this area, and giving a total protected area of nearly 100km^2 (J.A. Bohnsack pers. comm.) Before closure, there was intensive commercial and recreational fishing effort in the area and fish stocks were heavily exploited. Between 1957 and 1962, an average of 2.7 million kilograms of fish were landed annually in the vicinity of Merritt Island by 628 commercial fishers, and a further 1.47 million kilograms landed by an average of 764,000 sport fishers (Anderson and Gehringer 1965).

Using experimental fishing with trammel nets, Johnson et al. (1999) studied the fish populations in fished and unfished areas of the Refuge between 1986 and 1990 after 24-28 years of protection. They caught more fish and bigger fish in unfished areas than in fished areas. There were particularly striking differences in the abundance and size of key game fish species with overall game fish CPUE in unfished areas 2.6 times greater than in fished areas. The biggest difference in catch rates was for black drum, *Pogonias cromis*, which was 12.8 times more abundant in unfished compared to fished areas. Red drum, *Sciaenops ocellatus* was 6.3 times more abundant, the common snook, *Centropomus undecimalis*, was 5.3 times more abundant and spotted seatrout, *Cynoscion nebulosus*, was 2.3 times more abundant. The median and maximum size of red drum, spotted sea trout and black drum were significantly greater in unfished areas compared to fished areas. There was no significant difference in the median size of common snook between fished and unfished areas. Johnson et al. 1999 looked at other possible influences on these catch rates, such as habitat, and other environmental factors but found that fishing status was the most important.

Bohnsack (in Roberts et al. 2001) looked at the effect of these reserves on the adjacent recreational fishery by examining the frequency of world record size fish caught in proximity of the reserves compared to areas elsewhere in Florida. He looked at an area extending 100km north and south of the reserves (the distance being defined on the basis of tagging studies of movements of gamefish species) and found that though this only accounted for 13% of the Florida coast, 62% of record-size black drum, 54% of red drum and 50% of spotted seatrout were caught in this area. However, only 2% of record common snook were caught within the adjacent area. The first three game fish species are year-round residents of the Refuge, whereas snook is at its northern limit in the Refuge and was found to in winter (Johnson et al. 1999).

On closer examination of the occurrence of record size fish in the vicinity of the refuge, lag times were apparent from the beginning of protection to the point at which record fish began to appear frequently for each species. This is because it takes time for fish to grow large enough to exceed existing world records, and lag times were related to the longevity of the fish species (Figure 1). For spotted sea trout, which live for 15 years, records began appearing after 9 years; for red drum, which live for 35 years, the lag time was 27 years; and for black drum, which reach 70 years old, the lag time was 31 years. In the early 1980s new line classes were introduced and world records increased statewide as fish were caught for these line classes. However, adjacent to Merritt Island, only the shortest lived species, spotted seatrout, had grown sufficiently large by this time to show a spurt of new world records for these line classes.

By the end of the 1980s the rate of accumulation of new records slowed down for the shortest-lived species, the spotted sea trout. However, for the two longer-lived drum species, records continued to accumulate and since the mid 1980s most Florida records for both of these species have been recorded from the vicinity of the Merritt Island refuge (Figure 1). Captures of world record sized fish around the Refuge act as an indicator that spillover is occurring. However, the reserves would have been supplying smaller sized fish to the fishery for years before record-size fish appeared. This is confirmed by tagging studies.

Stevens and Sulak (2001) studied the movement of game fish in the Merritt Island National Wildlife Refuge. They caught fish in the closed areas from 1986 to 1988 and 1990 to 1992 and tagged the fish. Commercial and recreational fishers recaptured fish in the fishing grounds and researchers recaptured fish in the closed areas. They found that common snook migrated out of the protected areas south. This species was recaptured outside the closed area most frequently in their study, with 16.1% of the fish that were tagged being caught, compared with 0.8% for spotted seatrout, 2.9% for black drum and 3.1% for red drum. Snook were mainly recaptured far from the reserve, whereas the other species were caught mainly in the vicinity of the reserve. Common snook

moved the greatest mean distances (148km (±12.2km), followed by red drum 47.6km (± 6.6km), black drum 44.7km (± 18.2km) and spotted seatrout moved a mean distance of 10km (± 2.4km). Stevens and Sulak (2001) conclude that the closed areas do contribute large fish to adjacent fishing grounds and that red drum, black drum and spotted seatrout all spawn within the restricted area and potentially enhance juvenile recruitment to adjacent fished areas.
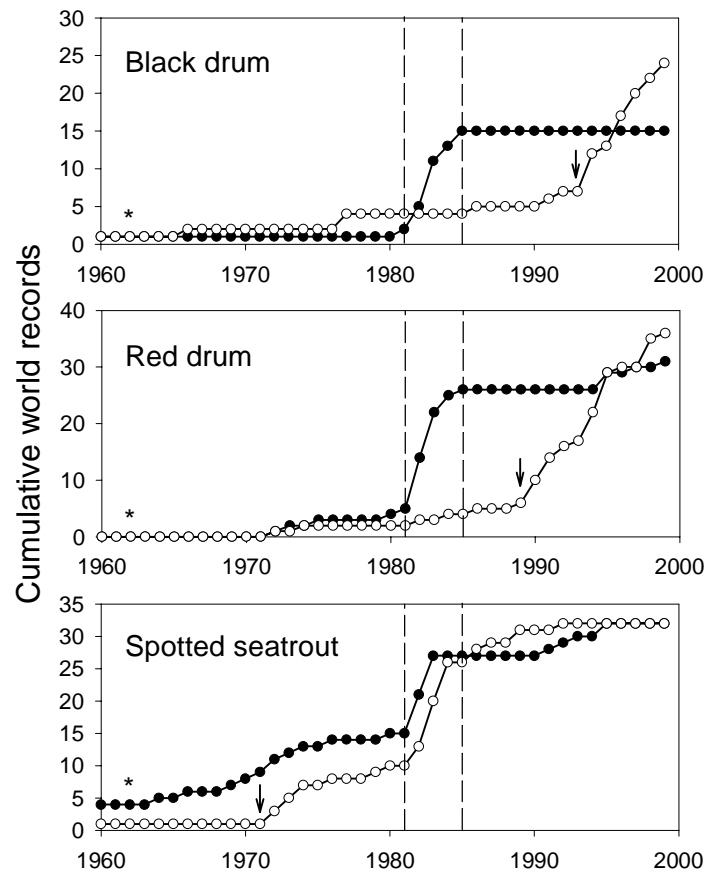
Figure 1: Cumulative world records for black drum, red drum and spotted seatrout in the 200 km coastal section adjoining the Merritt Island Wildlife Refuge (open circles) and records from rest of Florida (filled circles). Asterisks show date protection began. Dashed lines show period following introduction of new fishing line classes for records during which new records were added more quickly. Arrows show the points at which there was a rapid increase in rate of accumulation of new records for each species adjacent to the Refuge. Reprinted with permission from (Roberts et al. 2001). Copyright (2001) American Association for the Advancement of Science.

In a different study on the dispersal of oxytetracycline tagged juvenile red drum (the chemical leaves a mark in the ear bones of fish) released in an estuarine area in South Carolina, Collins et al. (2002) found substantial dispersal of fish but many remained within the area of release (1.8 by 3.2km) until they reached maturity at around three years old. After this they left the estuary. Collins et al. (2002) argue that their study supports the use of networks of small no-take reserves protecting the appropriate combination of juvenile habitat to enhance recruitment of red drum to the spawning stock.

Some critics suggest that the high numbers of world records from the Merritt Island area should be attributed to superior habitat in the area studied rather than to the closed status of the area (e.g. Wickstrom 2002). However, large areas of similar habitat can be found throughout Florida, for example in other areas of the Indian River Lagoon, Pensacola, St Andrews, Apalachicola, Tampa, Biscayne and Florida Bays, Charlotte Harbour, Ten

Thousand Islands, St Lucie canal and at the entrance to various other rivers. They do not have concentrations of world-record fish associated with them. The second largest concentration of record fish was associated with the Everglades National Park. This park introduced bag limits and minimum sizes before the rest of Florida. It also established a no-take area in 1980 to protect crocodiles and all commercial fishing in the area stopped in 1985. Using underwater visual census, Faunce et al. (2002) found that grey snapper, (*Lutjanus griseus*) an important recreational fishery species, were larger in the Everglades National Park Crocodile Sanctuary than in comparable unprotected area of Florida Bay and Biscayne Bay. They found that the modal size in the crocodile sanctuary was 25-30cm, whereas in Florida Bay and Biscayne Bay it was 15-20cm, below the minimum legal size for grey snapper in the recreational fishery, which is 25.4cm. In the crocodile sanctuary 66.2% of fish were above the legal capture size, whereas in Florida Bay 29.6% where legal and in Biscayne Bay just 15.6% were above the legal size. The crocodile sanctuary contains mangrove habitat and there are few such areas protected in the US, despite being an important habitat in the life cycles of many commercial fish. The use of marine reserves in the management of estuarine fisheries is still in its earliest stages around the world but this study provides convincing evidence that reserves work in estuarine habitats.

Another criticism levelled at the Merritt Island study is that world-record size fish are attributable to other management measures introduced in Florida, including bag limits, size limits and a ban on gill nets implemented in 1995 (e.g. Tupper 2002). However, such measures cannot account for the concentration of world record fish around the Refuge as they were applied state-wide and late in the study period. For example, the net ban was introduced state-wide years after catches of record-size fish began accumulating near the refuge. While such measures should produce greater catches of fish, it would take many years for fish to grow large enough to produce world-records, and such records would be expected state-wide. After the net ban 18 of 20 new records caught in Florida between 1996 and 1999 were from the vicinity of the Merritt Island Refuge. It is hard to think of more robust evidence for spillover.

Key points

- Marine reserves can work for game fish and benefit recreational fisheries.
- Marine reserves can benefit estuarine fish that move relatively long distances throughout life.
- Reserves at Merritt Island provide robust evidence for spillover, and probably also enhance reproduction and subsequent recruitment to the fishery.
- Reserve benefits can continue to build for long periods, and long-lived species may respond more slowly than short-lived animals.

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16. Gear closures in Britain

Britain does not have a history of using no-take marine reserves for either conservation or fisheries management. It has just two Statutory Marine Nature Reserves (SMNRs), although legislation has existed for their designation since 1981 (Jones 1999). There are no legal no-take marine reserves in UK waters, although the first is now being planned in Lundy Island SMNR off the coast of southern England. However, there are a number of restricted areas for mobile fishing gears and we look at the effects of two of them here.

Isle of Man closed area to mobile gear

In 1989 a 2km^2 area off the southwest coast of the Isle and Man, in the Irish Sea, was closed to trawling and dredging. The area was closed for research purposes by staff from the nearby Port Erin Marine Laboratory. The surrounding area is an important fishing ground for scallops, *Pecten maximus,* and prior to closure the area was intensively fished for scallops. The area around the closure is still one of the most heavily dredged in the Irish Sea (Bradshaw et al. 2001).

Since mobile gear was excluded, researchers have monitored the effects of protection from dredging using underwater visual transects of scallops and dredge and grab samples. Bradshaw et al. (2001) compared scallop populations and bottom communities with those in adjacent fished areas, and with areas within the closed area that were experimentally dredged. They found that scallop populations increased dramatically in the closed area from less than 2 per 200m^2 in 1989 to nearly 15 scallops per 200m^2 in 2000. Scallops also increased in the fished area from approximately 2 per 200m^2 in 1989 to approximately 10 scallops per 200m^2 in 2000. Scallop numbers were consistently higher in the protected area compared to the unprotected area, and scallops in the protected area were larger and older than those in the fished sites. In 1999 after ten years of protection, the mean age of scallops inside the closed areas was 6.5 years compared to 5.3 years outside. In the closed area the modal age categories were 6 and 7 years, whereas for the fished area modal ages were 4 and 5. Data presented by Bradshaw et al. (2001) from a study of the age structure of the local scallop population at the beginning of the scallop

Fishing boats in the Isle of Man.

fishery (Tang 1941) give an indication of the rate of recovery of the population. At that time the average scallop was 9.9 years old. A similar result was found for an area of the Skomer Marine Nature Reserve in Wales which

is closed to scallop fishing. Ten years after the designation of the marine reserve, densities of scallops had increased nearly four fold (Lock and Newman 2001).

Other species also appeared to be increasing in the Isle of Man closed area, including the starfish *Luidia ciliaris*, hermit crabs, spider crabs and brittlestars. Bradshaw et al. (2001) also found that that in terms of fauna found in grab samples, undredged plots were more heterogeneous than dredged plots, suggesting that dredging reduces heterogeneity in benthic communities, whilst protection increases diversity. Undredged plots were typically more structurally complex than dredged plots. The animals found most in dredged plots were encrusting species of sponge and bryozoans and small ascidians, all existing close to the substrate, whereas those found most in the undredged plots were upright species of bryozoan and hydroids, with much more three-dimensional forms.

Start Bay inshore potting agreement, south Devon

An area off the coast of Devon was closed to mobile gear as a result of a voluntary agreement initiated by the fishers themselves (Kaiser et al. 2000, Blyth et al. in prep). In Devon, fishers use pots to fish for crab (*Cancer pagarus*) and lobster (*Hommarus gammarus*), scallop dredges to catch scallops (*Pecten maximus*), and beam trawls and otter trawls to catch plaice (*Pleuronectes platessa*) and soles (*Solea solea*). This combination of gear types brings users into conflict because gears such as dredges and trawls either can't fish in areas where static gears have been set, or do fish there but remove or damage the gear. To manage this potential conflict, the fishers established areas of gear restrictions. These became formal in 1978 when a voluntary Inshore Potting Agreement was established. Kaiser et al. (2000) found that areas closed to mobile fishing gears (either year-round, or seasonally) had significantly different benthic communities to areas dredged or trawled. Communities in the closed area had higher biomass and more emergent fauna rather than in-fauna. An increased level of emergent fauna leads to increased habitat complexity in those areas too. Smaller animals and scavengers dominated areas open to towed gears. Highest species diversity was found in areas closed year round to mobile gear, followed by areas closed seasonally to mobile gear, followed by areas used by mobile gear. Biomass of soft corals and hydroids was higher in areas that were closed to towed fishing gear. No data are available yet on how the differences in benthic communities might influence fisheries, but the main aim of the closure system, to reduce conflict between users of different gears, seems to have been achieved (Kaiser et al. 2000).

Key points

- Closed areas can lead to rapid rebuilding of exploited populations like scallops.
- Protecting areas of temperate water continental shelf can lead to habitat recovery from damage by mobile fishing gears, allowing development of more biologically rich and complex habitats.

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