Review

Challenges for Managing Fisheries on Diverse Coral Reefs

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Abstract: Widespread coral reef decline has included the decline of reef fish populations, and the subsistence and artisanal fisheries that depend on them. Overfishing and destructive fishing have been identified as the greatest local threats to coral reefs, but the greatest future threats are acidification and increases in mass coral bleaching caused by global warming. Some reefs have shifted from dominance by corals to macroalgae, in what are called "phase shifts". Depletion of herbivores including fishes has been identified as a contributor to such phase shifts, though nutrients are also involved in complex interactions with herbivory and competition. The depletion of herbivorous fishes implies a reduction of the resilience of coral reefs to the looming threat of mass coral mortality from bleaching, since mass coral deaths are likely to be followed by mass macroalgal blooms on the newly exposed dead substrates. Conventional stock assessment of each fish species would be the preferred option for understanding the status of the reef fishes, but this is far too expensive to be practical because of the high diversity of the fishery and poverty where most reefs are located. In addition, stock assessment models and fisheries in general assume density dependent populations, but a key prediction that stocks recover from fishing is not always confirmed. Catch Per Unit Effort (CPUE) has far too many weaknesses to be a useful method. The ratio of catch to stock and the proportion of catch that is mature depend on fish catch data, and are heavily biased toward stocks that are in good condition and incapable of finding species that are in the worst condition. Near-pristine reefs give us a reality check about just how much we have lost. Common fisheries management tools that control effort or catch are often prohibitively difficult to enforce for most coral reefs except in developed countries. Ecosystem-based management requires management of impacts of fishing on the ecosystem, but also vice versa. Marine Protected Areas (MPAs) have been a favorite management tool, since they require little information. MPAs are excellent conservation and precautionary tools, but address only fishing threats, and may be modest

fisheries management tools, which are often chosen because they appear to be the only feasible alternative. "Dataless management" is based on qualitative information from traditional ecological knowledge and/or science, is sufficient for successful reef fisheries management, and is very inexpensive and practical, but requires either customary marine tenure or strong governmental leadership. Customary marine tenure has high social acceptance and compliance and may work fairly well for fisheries management and conservation where it is still strong.

1. Introduction

Coral reefs have been widely reported to have declined substantially around the world [1–39], and recent reports have provided quantitative evidence of that decline (e.g., [40–47], but see [48–50]). So far, great efforts have not stopped the decline, though the reefs might have been worse without those efforts [22]. Coral reefs are shallow marine ecosystems that build geological structures by depositing calcium carbonate produced by corals, algae, and various other organisms. Coral reefs are commonly said to be the most diverse marine ecosystem. Coral reefs actually vary widely in diversity, from a high in an area of islands near Southeast Asia known as the "Coral Triangle" to lows at the northern, southern, and eastern edges of the Indo-Pacific and in Brazil [51].

Coral reefs provide important ecosystem services to humans, such as food, shoreline protection, tourism, and many others (e.g., [52,53]). The total estimated ecosystem services of coral reefs worldwide is considerable, over US$375 billion per year [42], over five times larger than the total value of all the world's marine fisheries (US$70 billion), most of which is not from coral reefs. Around the world, about 850 million people live within 100 km of a coral reef and 275 million live within 10 km of the coast and 30 km of a reef [54], over 91% of the people living within 100 km of a reef are in developing countries [55], and many of them are poor and depend on reef fish for their primary source of protein. Many of the world's fisheries are overfished or recovering from overfishing (e.g., [56–67]), and the emphasis now is often on rebuilding fisheries (e.g., [68–70]). There are fisheries that are overfished, some are fully exploited, some that are not fully exploited, and others are being rebuilt. One study reported that 55% of island countries are overfishing their reefs and total landings are 64% higher than can be maintained [26]. Pandofi *et al.* [41] reported that fishing had major effects on coral reefs long before other local human effects. "Of the numerous threats to biodiversity, fishing is arguably the most pervasive and damaging" [71]. Reefs at Risk Revisited [54] ranked overfishing and destructive fishing as the greatest local threat to coral reefs globally, Roberts [72] stated that "By virtue of their complexity, reef fisheries are the most difficult in the world to manage," Sale [18] stated that "… most reef fisheries are unmanaged or undermanaged …" and Munro [73] stated that "Reef fisheries have a dismal management record." There is evidence that at least one coral reef fish (bumphead parrotfish, *Bolbometopon muricatum*) has been driven to local extinction by fishing some places [74]. The largest giant clam, *Tridacna giga*s, has been driven to local extinction in many islands of

Micronesia, Vanuatu, and probably New Caledonia, and certain reef fish species in parts of Micronesia and Polynesia [75]. Local extinction produced by exploitation is a clear management failure.

Coral reef fisheries lie at the intersection of at least three quite different disciplines, coral reef ecology, fisheries science, and social science. Individuals are usually trained in just one of these. Projects working on coral reef fisheries need teams that include all three. Fisheries and ecology have different values and goals, with conservation the goal of most ecologists, and one kind of ecosystem service being the goal of most fisheries scientists. These goals or values are the source of different views on many matters, and sometimes leads individuals to accept evidence that is consistent with their values less critically than evidence that is not. It is very difficult to remain objective in these circumstances, and unstated values often color interpretations of facts. These different values can also lead to similar views, so for instance, much of fisheries science is directed towards limiting fishing pressure to conserve stocks so that they can continue to provide ecosystem services indefinitely.

2. Resilience

This first section is concerned with the possibility of a synergy between the effects of fishing and the effects of climate change in degrading coral reefs. Global warming is increasing the frequency and severity of mass coral bleaching, which is causing coral mortality. Healthy coral reefs can show resilience by recovering rapidly from mass coral deaths, but the loss of herbivores, including fish, may allow blooms of macroalgae on newly killed coral surfaces, impeding the recovery of coral communities. Thus, overfishing, particularly of the herbivorous fish that eat macroalgae, can exacerbate the effects of mass coral bleaching by slowing or stopping recovery. Living coral is better habitat for fish than rubble or algae beds, so the loss of living corals eventually leads to reductions of fish populations and fish catch, and thus ecosystem services.

2.1. Climate Change

Climate change, including mass coral bleaching, and acidification are now often considered the greatest future threats to coral reefs worldwide (e.g., [12,31,76,77]). Coral bleaching, where corals expel their symbiont dinoflagellate zooxanthellae (*Symbiodinium*), can be produced by a variety of stresses, but is primarily caused by high water temperatures, and mass coral bleaching can be predicted quite well with sea surface temperatures [78–83]. Corals can recover from bleaching if it is not too intense, but intense bleaching causes coral death. In the El Niño of 1998, 16% of the world's coral is estimated to have died [84], the largest single coral mortality known. Global warming is predicted to continue to increase sea surface temperatures globally, and increasing ocean temperatures will mean that years when events such as El Niño produce temperatures that are high enough to cause mass coral bleaching and mortality will occur more and more often. Within a few decades, bleaching is predicted to occur annually and mortality events will have too little time between them for recovery [19,85], though if corals can adapt that would be delayed significantly [86]. Annual summer mass bleaching of multi-species coral communities has already begun in at least one location [87]. Lower temperatures, shading, and water currents [88] may be able to reduce bleaching. Unfortunately, none of these have been shown to be practical at anything other than very small scales, leaving managers with no options

for directly combating bleaching in local areas. International agreement to reduce greenhouse gas emissions sharply is critical to stop climate change and acidification

2.2. Reef Resilience

The primary tool for battling coral bleaching that managers are left with is to increase reef resilience. Resilience is generally defined as the ability of an ecosystem to recover from disturbance, or the rate at which the ecosystem recovers (e.g., [89–91]). This can be called "engineering resilience" and contrasted with ecological resilience which recognizes tipping points between alternate stable states maintained by self-reinforcing mechanisms [92]. Increasing resilience can help buy time while the causes of global warming (the emissions of greenhouse gases) are reduced. In general, it is presumed that healthy ecosystems will be able to recover from disturbances. Coral reefs have survived disturbances such as cyclonic storms for their entire existence. Reefs that are exposed to cyclonic storms every few years are likely to have recovered from hundreds such storms in the Holocene, and thousands in the entire history of the geological structure. But the typically chronic harmful activities of humans can reduce the rate at which reefs recover from natural disturbances. For instance, the Great Barrier Reef has been reported to recover more slowly from more recent disturbances such as crown-of-thorns outbreaks than earlier disturbances [93,94], and declines in cover have been suggested to imply reduced resilience ([48] but see [49,50]). Coral cover has decreased from a mean of 28% to 22% over 19 years, but decreases were in localized areas, while most reefs did not decline [47]. *Montastrea annularis* colonies in the Caribbean exposed to local stressors had growth slowed for at least eight years after a bleaching event, while colonies not exposed to stressors recovered normal growth rates in 2–3 years [95], and stressors increase the ability of degree heating weeks to predict slowed growth [96]. To maximize resilience, all human activities damaging coral reefs need to be reduced and minimized [97]. That includes actions such as overfishing and destructive fishing, sedimentation, nutrient runoff, other pollutant runoff, and the myriad of other harmful effects humans have on coral reefs.

Fishing may reduce coral reef resilience in several ways. First, there is now a published report that reefs in No-Take Areas (NTAs) have fewer outbreaks of crown-of-thorns starfish (COTS) than outside NTAs on the Great Barrier Reef [98]. Crown-of-thorns sea stars kill coral, and surely coral mortality from outbreaks make the recovery of a reef from mass coral bleaching even more difficult. A frequently cited report that COTS are more common when fishing pressure is lower [99] can also be explained by nutrients fueling plankton blooms that provide food for starfish larvae, since population was used as a proxy for fishing pressure and populations produce nutrients [100], a hypothesis with strong support [101]. Second, Marine Protected Areas (MPAs) in the Philippines have been shown to have less coral disease. MPA protection usually increases the number of larger predatory fish, which in turn can reduce the number of smaller fish such as butterflyfish. Some butterflyfish eat coral, and interestingly, only the density of butterflyfish that eat coral correlates with the abundance of coral disease. The butterflyfish are probably transmitting disease from coral to coral by feeding on diseased coral and then healthy coral [102]. Third, coral cover declines less rapidly or recovers better inside MPAs than outside [103,104]. Fourth, fishing on coral reefs has greatly modified the fish community. In particular, on reefs anywhere near humans around the world, most of the largest fishes, such as

sharks, humphead wrasse (*Cheilinus undulatus*), bumphead parrots, and goliath grouper (*Epinephelus itajara*) as well as other megafauna such as sea turtles and monk seals, have been removed, many long ago [9,32,42].

2.3. Apex Predators

The abundance of apex predators on natural reefs was not appreciated until a series of recent reports from remote near-pristine reefs in the Pacific [16,23,30,32,105–109]. The correlation between the abundance of the largest fish species and fishing pressure is strong [23,38,110,111]. The largest species are at low abundances nearly everywhere people are present [10,32], the acceptance of which is an example of "shifting baselines" [112,113]. In Fiji, bumphead parrots filled fish markets on some islands when night spearfishing was introduced, and now the species is locally extinct on some islands [74]. It is well known in fisheries generally that fishing removes the largest fish first (e.g., [68,114–120], Figure 1). "Some of the larger slower growing species can become very vulnerable to modern gears and be fished to extinction" [75]. All of the effects of removing the largest reef fish species are not yet clear; most of the world's reefs have had these fish removed, yet major effects on those reefs have not been documented. The removal of top predators often produces a chain of effects called a "trophic cascade" where the removal of a predator leads to an increase in its prey, which in turn produces a decrease in that species' prey and so on [121–123]. In many cases the results are dramatic, but they appear not to be on coral reefs very often. Two cascades are known from the Indo-Pacific, one in Kenya where triggerfish eat urchins which eat algae [124] and urchins cause larger decreases in crustose coralline algae than fish so that fishing causes increases of urchins which cause decreases of both fleshy algae and CCA [124,125] and another on the Great Barrier Reef where a line fishery for one species of grouper causes increases in abundance of prey species [126]. Three in the Caribbean, one where herbivorous fish and urchins eat algae [127], and another where predators eat small parrotfish but not large parrotfish and the large parrotfish eat algae [128] and a third where large groupers cause smaller groupers to hide and grow slower, and new recruits of other species have higher survival rates, presumably from lower predation by the small groupers [129]. Fishing removes medium and small fish as well as large fish, so it may block trophic cascades [130,131].

The change from an apex predator-dominated community to a community with few apex predators is itself a large phase shift, the most widespread phase shift known on coral reefs by far. Only about 4% of Caribbean reefs and 1% of Indo-Pacific reefs have high macroalgae cover indicating they have undergone a phase shift from coral to algae [132], though the criterion of 50% algae cover in that study has been criticized as too high [126]. The apex predator loss also qualifies as ecosystem overfishing, which is when fishing changes community composition [115].

Figure 1. Humphead wrasse (*C. undulatus*) and the fisherman who caught it in American Samoa. This fish sold for $120. A similar fish in Guam would now sell for about $2,000. Photo by Leslie Whaylen, 2005.

2.4. The Role of Herbivores

The best studied example of fishing reducing coral reef resilience comes from the removal of herbivores. In Jamaica, Hurricane Allen killed large amounts of coral in 1980 [133] and was followed in 1983 by the reduction of the population of the sea urchin *Diadema antillarum* by two orders of magnitude by an unknown disease which swept through the entire Caribbean over the course of a year [134]. This was followed by a bloom of macroalgae and then further loss of coral [7], which has persisted to this day on most of the reefs. This has been called a "phase shift" from corals to algae,

since coral cover plummeted as the macroalgae bloomed. Phase shifts have been defined as significant changes in community structure and composition (alternate state) [92]. Phase shifts to dominance of other organisms have also been documented [135–137], and indeed the loss of mobile megafauna or apex predators from coral reefs should be considered a phase shift itself. The reefs of Jamaica had been heavily fished for at least 100 years [33,60], and were heavily overfished at that time, with most of the few fish to be seen on the reef already in fish traps. This included the herbivorous fish. The sea urchins were the last remaining herbivores, and once they were gone and new substrate was opened up, there was nothing to restrain the growth of macroalgae [7]. Some have concluded that the loss of herbivores led to the growth of macroalgae, which in turn killed the coral. In Jamaica, Hurricane Allen caused most of the coral mortality, and across the Caribbean disease killed most of the coral, primarily white band disease killing *Acropora*. Algae then colonized the dead surfaces [131]. Much of the decline in coral cover happened before the *Diadema* dieoff [40].

2.5. Nutrients

Nutrients may have played a role as well [2,138–142], and runoff had a greater effect than herbivores reduced by fishing, on macroalgae in a meta-analysis of studies in the Caribbean [143]. Top-down control by herbivory has had the most support (e.g., [7,25,144–146]) and indeed is easy to demonstrate. If an area of reef is caged off, so herbivores are excluded, dense growths of macroalgae generally result, and when the cage is removed, herbivores eat the macroalgae [24,147–149]. Herbivory, nutrients, algae and corals are involved in many complicated interactions [142,149–157] which cannot be reviewed here, but clearly herbivory is one of the important factors influencing the abundance of algae.

2.6. Phase Shifts and Herbivores

Very few coral reefs that have had a phase shift from coral to macroalgae have ever fully recovered, though the macroalga that dominated Kaneohe Bay, Hawaii, has greatly decreased [158], and there are some signs of *Diadema* recovery and local decreases of algae and coral recovery in the Caribbean [159–162]. Phase shifts that do not revert to the original state may imply that coral reefs can have alternate stable states. Phase shifts to alternative stable states are referred to as regime shifts by some [92]. Environmental events or pressures may move a reef from one state to another, but once there the reef stays there unless conditions push it into the other state [163]. Another idea is that there may be hysteresis, positive feedback effects, or a ratchet-like mechanism such that the transition from coral to macroalgae occurs at one level of herbivory, while the return to coral requires a higher level of herbivory [164–166]. Nyström *et al.* [92] list eight possible positive feedback effects. For instance, the number of herbivores needed to control algae on a reef dominated with coral and little macroalgae should be much less than the number of herbivores needed to remove macroalgae from a reef where a disturbance like mass bleaching mortality has produced a large area of bare substrate which has been colonized by macroalgae, so that there is now low coral cover and orders of magnitude more macroalgae than when the reef was dominated by corals. There might even be too much macroalgae for a pristine herbivore community to control, let alone remove [13,167–169]. There is experimental evidence that supports this idea [170]. Macroalgae do not provide as good habitat for fish as corals, and may lead to declines in fish [25,171]. Once corals have died, they will eventually collapse,

reducing rugosity and hiding places and leading to further reductions in fish populations [45]. Both the increase in macroalgae and the loss of rugosity can further reduce the abundance of herbivorous fish, making it harder for the reef to recover. The implication of these ideas is that it may be much easier to avoid a phase transition from coral to macroalgae than to reverse it. However, few reefs that have undergone phase shifts have had the causative factors removed, so it is not clear whether they would recover on their own. One possible example, though, is a protected area in Chagos which lost most of its coral cover in the 1998 mass bleaching. It had good fish abundances, and macroalgae was kept low, but corals have not returned [172]. There may have been no remaining source of coral larvae, showing that if the destruction of coral is widespread, loss of coral recruitment can greatly retard recovery. A meta-analysis found that recovery was slower when the nearest reef was farther away. Recovery was also faster when coral cover was low after disturbance, faster after COTS or mixed causes than bleaching or storm, and differs by ocean area [173]. A model indicated that continuous shifts are more likely than discontinuous shifts, and reducing several human stressors at once increases resilience more than single stressor reduction [97]. Another model indicated that warming and acidification produced by increased CO_2 in the atmosphere reduces reef resilience, and overfishing and nutrient loading made reefs more subject to the effects of CO_2. Further, at future CO_2 levels, controlling such impacts will be crucial to retaining coral populations [174].

Not all herbivores are equal, and in particular, several functional groups of herbivorous fish have been distinguished based on their feeding mode. The majority of herbivorous fish eat little or no macroalgae. Herbivores may remove newly recruited macroalgae as they scrape or excavate turf [175], and the bases of macroalgae which slough off their top portions seasonally [176]. There may be a need for a balanced community of herbivores with representation from each of the functional groups to control macroalgae, and perform other functions for coral reefs [169]. Turf is eaten primarily by smaller herbivorous fish, and the species that control macroalgae tend to be larger, and so more readily removed by fishing. Excavators remove dead corals [21] which commonly attract coral recruitment, but when the corals eventually collapse, the juvenile corals are left without firm attachments and are thus unable to survive. Size is important for the ability of parrotfish to remove algae by excavating and scraping, large individuals are able to do more [92]. While the excavator functional group in the Pacific consists of several species [177], a few species account for most of the bioerosion, with the most often produced by bumphead parrotfish [110]. This species is by far the largest parrotfish, which is easily extirpated by night-time spearfishing, since they sleep in schools at the same location each night where they can easily be speared [74]. The fish species that keep macroalgal abundances low on a healthy coral reef may not be the same species that remove macroalgae once they dominate [37,169]. When cages were removed in one experiment, the fish that ate the macroalgae was a species not even known to be an herbivore, the batfish *Platax pinnatus* [148]. In areas where it has been fished out, the reef may have reduced resilience, without anyone knowing it, just as Jamaica had reduced resilience before the phase shift without anyone realizing it. Even reefs that are not overfished by fisheries standards may have reduced resilience due to reduced herbivore populations [14]. Green and Bellwood [169] have provided a detailed rationale and methodology for monitoring functional groups of herbivores, and indeed it appears that for most monitoring programs the existing fish data can simply be categorized by functional groups. Functional categories are not always simple, for instance many herbivorous fish are carnivorous as juveniles, detritivores often dominate reef fish biomass yet

are often lumped with herbivores [178], some parrotfish feed on corals as well as algae [110,179], and some fish take a variety of foods. Several authors have pointed out that studies of function and process on reefs are needed in addition to basic monitoring [21,165,169,180]. Phase shifts are an example of an ecosystem effect of fishing; the need to manage such effects is now widely acknowledged [181] (Section 4.3). Avoiding phase shifts has been proposed to be a prime goal in coral reef management [21,92].

The loss of living corals and replacement by macroalgae, leads to changes in the structure of reef fish communities. The death of living corals primarily impacts corallivores and fish that live in the branches of corals, which depend directly on living corals [182–184]. These fish tend to be small and so less important for fisheries. Dead branching and plate corals eventually collapse from bioerosion, reducing rugosity and hiding places for fish. Loss of structural complexity reduces the carrying capacity for fish [185–187]. It can take five years for corals to collapse, so there is a lag before the effects on fisheries are likely [188]. This then is more likely to produce reductions in a broader range of fish species [189–191], though a few species may increase greatly [189]. Loss of coral can also lead to loses in fish diversity [187,191]. In addition, macroalgae is generally a poor habitat for reef fish [46]. The replacement of live corals by rubble or macroalgae is likely to lead to reductions in fisheries. Only 6% of species in reef catch are those that are lost immediately after coral death because most of such species are small, while 56% of species caught are those that are lost after habitat loss [192].

To summarize, the main lines of evidence that support the theory that insufficient herbivory contributes to phase shifts from coral to algae are a correlation of algae abundance with fishing pressure or herbivorous fish abundances [130,193], observations of macroalgae increases following herbivore exclusion and decreases following removal of exclusion, and modeling studies. Experiments using caging and transplantation have shown that herbivores can both prevent and reverse phase shifts in very small areas [193] (and references therein). A model indicates that good fisheries management can delay the sustained loss of corals from reefs experiencing periodic mass coral bleaching by 18–50 years [194].

Although the concepts of resilience, phase shifts, and hysteresis appear to be widely supported in the coral reef ecology community, problems remain. The loss of corals is widely documented, increases in algae reported, and many reefs have not recovered. Still, phase shifts are not as common as often thought [132] and resilience is not well enough defined for some. Resilience can't be measured directly, the only sure way of knowing where a threshold is, is to cross it [195] and the evidence that there are hysteresis effects is slim. If the original cause of the change in community structure has not been removed, the lack of recovery is not good evidence of a hysteresis effect. There is confusion in the use of terms. Dudgeon *et al.* [196] make a cogent argument that phase shifts are responses to persistent changes in the environment, regime shifts are the same thing, and phase shifts can be small or large changes in the community which are gradual or sudden. A phase shift is the path that the community takes from an old equilibrium point to a new equilibrium point. To reverse the phase shift it is necessary to reverse the environmental change. In almost all cases on coral reefs (including the coral to algae shift in Jamaica and the rest of the Caribbean), either the environment has not returned to its previous state or the return of environment has led (often slowly over decades) toward the recovery of the community. So, for instance, increases in *Diadema* populations have led to decreases in algae and increases in coral [159–162,196]. Just as fish populations usually recover if

fishing stops (but for some species the recovery may be very slow), reefs can recover from phase shifts if the causes are removed, such as the lack of urchin herbivory. Alternative stable states occur only if the community does not return to the original state when the environmental change is removed. Contrary to common views, phase shifts can be hard to reverse if the environmental variable is hard to reverse, and alternative stable states need not be hard to switch between if the attraction surface is shallow [196]. Dudgeon *et al.* [196] review seven examples of phase shifts on reefs to examine the evidence for alternative stable states, and conclude that only one may be an alternative stable state, flattened substrates after ship groundings. Loose rubble beds can also be persistent [197] and might be an alternative stable state. Dudgeon *et al.* [196] did not find evidence of hysteresis. It may be difficult to distinguish alternative stable states from very slow recovery. Better ways of testing the concepts of resilience, alternative stable states and hysteresis on coral reefs are needed.

3. Fisheries Stock Assessment and Diversity

3.1. Conventional Stock Assessment

Fisheries management, like medicine and car repair, can be considered to consist of four main steps, determining if there is a problem, determining the cause of the problem, choosing tools to correct the problem, and using the tools to correct the problem. Low fish catches indicate a problem, and low abundances of large fish could also suggest a problem. Oceanography could indicate whether low productivity contributes to the problem, ecology could indicate whether habitat degradation contributes, and other disciplines could contribute as well. The "gold standard" for determining whether too many fish are being taken (overfishing) or the fish are depleted (overfished stocks) is "stock assessment". This is based on mathematical models of fish stocks, reproduction, natural mortality, and fishing pressure [198]. Fish stocks are similar to populations; they are groups of fish that are largely independent from other groups of the same species [198]. So the population of a reef fish species around a particular small island that is isolated by large distances from other islands is a fish stock. The fate of the fish around that island is almost completely independent in the short to medium term from those around other islands. For some species, even individual reefs might have separate stocks. It is commonly said that most reef fish have a larval dispersal stage, but rarely noted that elasmobranches which compose around 50% of the fish biomass on near-pristine reefs, have no larval dispersal phase. Adult reef sharks of some species have larger home ranges than most other reef fish. Note that a stock is a subset of a species, but that coral reefs have highly diverse reef fish communities with around 250–2,000 fish species in an archipelago [199]. Reef fisheries take about 200–300 species in the Pacific and about 100 in the Caribbean, but usually fewer than 20 species make up more than 75% of the weight of the catch [73]. That might suggest that the fishery is not as complex as it appears [200]. However, fish species that are in the worst condition (overfished, ecologically and economically extinct and nearly locally biologically extinct) will be so rare they will rarely if ever appear in the fish catch, let alone in the 20 most common species in the catch. The most abundant species in the catch are unlikely to be overfished, because if they were overfished they would likely not be abundant in the catch. Restricting the study to only common species in the catch means the findings will be reliable, but it highly biases the findings to only species that are in above average condition, and makes it quite
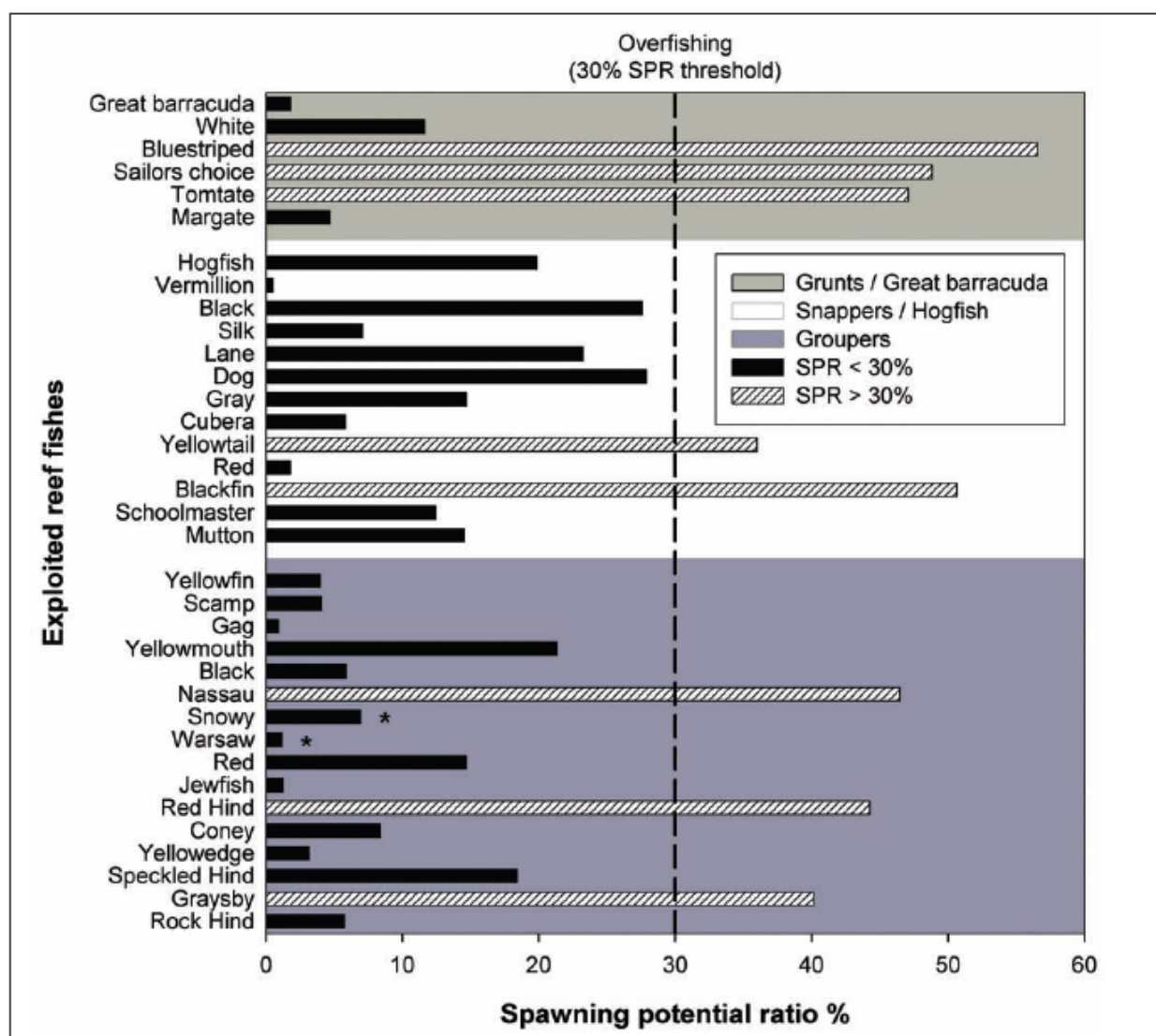
incapable of finding the species in the worst condition. All methods that depend on fish catch data are subject to this problem, and are nearly useless for reef fish conservation. Sustainability is often said to be the goal of conservation, but it is possible to sustain stocks and catches at low levels [201], and shifting baselines can trick us into accepting those low levels. A better goal is rebuilding overfished species [68–70]. Species that are rare in the catch cannot be ignored if rebuilding overfished species is a goal. Good management includes an element of triage, determining what the greatest problems are and setting their correction as the top priority.

Sometimes all reef fish are considered together as a group, so for instance the total catch of all reef fish in an area (e.g., [202–204]), or total biomass (e.g., [205]) is reported. If reef fish species are lumped together in management, there are huge risks of overfishing or even local extinction of some species [206,207]. Reef fish species have a wide range of attributes in many dimensions. So, for instance, in size they range from tiny gobies to giant groupers weighing up to 300 kg. There are a wide range of diets. Reproductive potential ranges from bony fish that spawn millions of tiny eggs that disperse, to sharks that have a few pups a year and have no larval dispersal stage. Reef fishes have a wide range of vulnerability to fishing [208], with the largest reef fish typically having vulnerabilities of about 70–80 on a 0 to 100 scale, to small fish with vulnerabilities around 25, and one of the most common Pacific reef fish (lined surgeonfish, *Ctenochaetus striatus*) having a vulnerability of 13 [209]. In addition, they vary greatly in their "resilience" or speed with which they can restore their populations [209].

In the few stock assessments that have been done on reef fish (e.g., [211,212–216]), each fish species was found to be in a different condition (Figure 2) and they are different at each location [211–214], (Figure 2). Overfishing is defined in fisheries for individual species (actually stocks within species) not groups of species, and a statement like "this reef is overfished" is an oversimplification (unless one is referring to ecosystem overfishing), as some species may be overfished while others are not. The result is that if you lump reef fish together and manage for maximum yield for small species that have low vulnerability and high resilience, medium size fish are likely to be overfished and large fish may be driven to economic, ecological, or even local biological extinction. Managing for maximum yield of the entire fish community as a whole is likely to lead to the same result. On the other hand, if you manage for maximum yield for the largest species, you will unduly restrict the catch of medium and especially small species, causing great hardship for poor fishers [198]. One can define a Multi-Species Maximum Sustainable Yield (MMSY) [205] but the maximum yield produced by managing the group of species [75] will be less than that which could be produced by managing each species separately for its own Maximum Sustainable Yield (MSY), since if they are managed as a group, many species will be either overfished or not fully fished; many or most will not be at MSY. One size does not fit all, and there are major risks for managing them all as a single group. MSY is now usually seen as a poor target for fisheries, due to uncertainties in calculating it [201], recruitment being highly variable in time and space [215], the fact that Maximum Economic Yield (MEY) is well below MSY, and other factors. MSY is often "incompatible with economically viable fisheries" [216]. Instead, Optimum Yield (OY) is often preferred, which is a point that has a precautionary safety margin built in, and which is closer to MEY. In an open-access system, fishers will continue to enter the fishery as long as they are able to profit more than in other occupations. The equilibrium point in such a system is at the point where each fisher makes no profit after costs are deducted, including opportunity costs of other

jobs not pursued. If alternative jobs pay little, that point is pushed far beyond MSY to where catches are very small and fish are heavily overfished. In such a situation, which describes a large part of world coral reef fisheries, people are pushed by desperation to "Malthusian overfishing" in which destructive fishing is often used [217], further reducing fish catch.
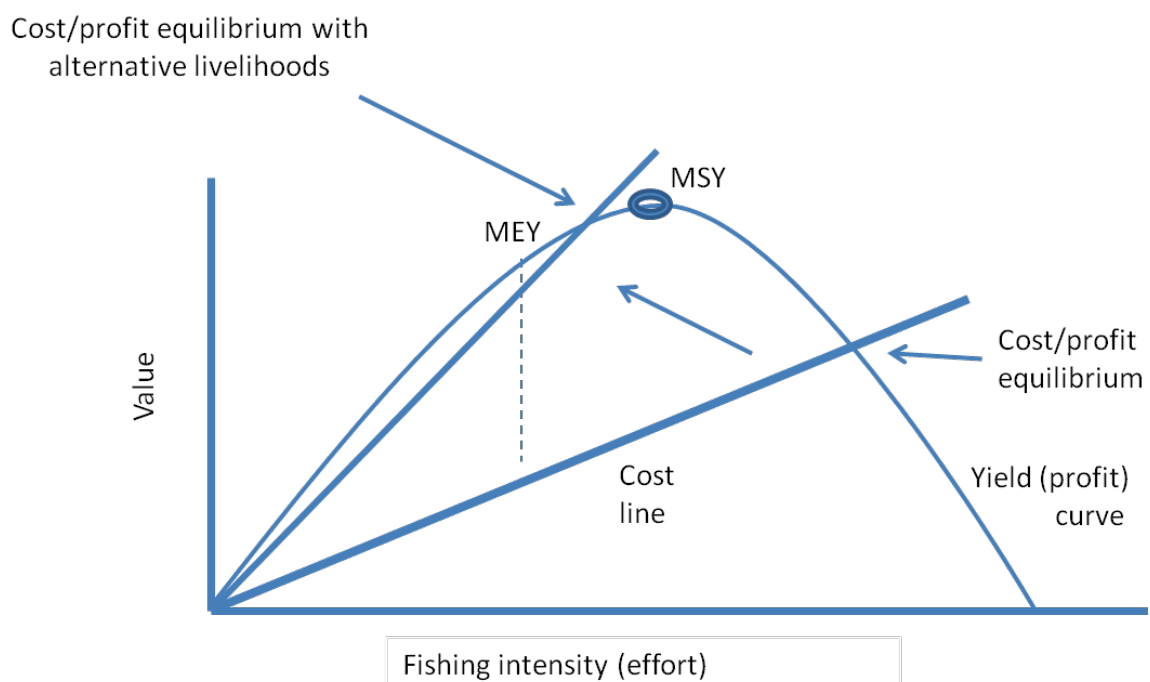
Figure 2. Spawning potential ratio for reef fish for 35 species from Biscayne National Park-Florida Keys. Bars that are less than 30% (black bars) indicate overfished species, those that are more than 30% (striped bars) are not overfished. Notice the wide variation from one species to another. Reproduced from [210], derived from [211].

MEY in this model (Figure 3) applies to commercial and subsistence fisheries, since profit or catch is the motive. For recreational fisheries, fishing is heavily or completely subsidized by the non-fishing income of the fishers. Thus, the cost line for recreational fishing is pushed to the right, and may even be at zero, and thus in theory could push stocks to biological extinction. However, recreational fishers are likely to find not only fishing intrinsically rewarding, but also catching fish, and catching large fish more than small fish (taking fish home is also surely rewarding). So they may actually be willing to have the take-home catch greatly restricted, in order to be able to catch more fish. Such is the case in

"catch and release fisheries," which do not make any sense for commercial fishing. Catch and release fishing can be common in areas with large recreational fisheries, for instance about half of the fish caught by recreational fishers in Florida are released [214]. In Florida, the annual catch of the estuarine fish called common snook (*Centropomus undecimalis*) is larger than the total stock, because of catch and release [218]. This catch and release is forced by a very narrow slot limit; fish outside the slot limits must be released. Recreational fishing on coral reefs can be very large in developed countries. It is much larger than commercial fishing in Florida [214] and Hawaii [219]. In Florida, in a two day recreational lobster fishing season, more than 50,000 divers remove about 80–90% of the stock each year [220,221]. However it is likely to be small in developing countries where only the wealthiest citizens could afford it.

Figure 3. Fixed-price model of a harvest system. The value axis is the cost of fishing (including opportunity cost) or the value of the catch and thus proportional to the catch. MSY = Maximum Sustainable Yield, MEY = Maximum Economic Yield. Redrawn from [217].

Fisheries models all have assumptions, and the assumptions are rarely tested. One in particular is important, that a stock will grow faster when it has been reduced by fishing than when it is unfished and abundance is higher [61,198]. This is essentially density-dependence. Not all reef fish or other fish have density-dependent populations (e.g., [222]). Fisheries models predict that a stock that is no longer fished will recover, but Hutchings [58] has documented 41% of 90 cases where they have not, including Canadian cod, a spectacular failure of fisheries model application and management [56,57] (but see [59]). There are several examples of pearl oyster populations that were severely depleted 50–100 years ago which have not recovered [75]. If stocks will not recover in any reasonable time, avoiding overfishing becomes even more imperative, like avoiding extinctions. Surplus-production fisheries models were designed for apex predators that had no predators preying on them; for other species, much of the surplus production is taken by predators [223]. Density dependence may be the rule

in model single species systems like bacterial cultures, but in complex ecosystems it is not guaranteed. Further problems occur in stock assessment as there is often great uncertainty in variables put into the models, leading to arbitrary decisions that are not apparent in the final figures produced [201].

The diversity of coral reef fisheries poses great challenges to fisheries stock assessment. Not only is the fish community highly diverse, but also there are a variety of invertebrates that are taken in fisheries (including coral), there are special sectors of the fishery, some of which are very large and all of which have their own problems (such as the live food fish trade [224], the aquarium fish and coral trades [225], invertebrate fisheries, cyanide fishing, destructive fishing [226], and spawning aggregations [227–230]). There is a wide variety of gear used, and there are tens to hundreds of millions of fishers spread through about 100 countries, one of which is the fourth most populous on the planet (Indonesia) with over 13,000 islands, and the second largest area of coral reefs of any country, just less than Australia [54]. The resources available are no match for the task. Stock assessment requires a great deal of biological information for each fish species (some models are now baroque in complexity, with hundreds of free parameters [61]), it requires a great deal of expertise, and it is very expensive. This includes multispecies virtual population analysis [231].

Compare coral reef fisheries with pelagic fisheries. Tuna is a tropical fishery with about 5 species around the world, and the fishery is worth about US$1.6 billion dollars a year in the Pacific alone [75]. The fishery is largely industrialized, with a single purse seiner capable of carrying about 1,000 tons of tuna. The fishery is also concentrated, with relatively few boat owners, few landing sites, and large cash flows in relatively few hands. Expensive, expertise-intensive stock assessments are cost effective for such a fishery. Contrast that to coral reef fisheries, where hundreds of fish species may be caught in any one area, there are millions of fishers spread out along very long coastlines with landings all along the coast, most fishing is subsistence or artisanal, very little money changes hands [224] but large numbers of very poor people depend on reef fish for their primary protein source [115,224], in low-income countries. The importance of reef fisheries is not in the amount of the catch or the cash generated, but in feeding tens or hundreds of millions of hungry people [115]. There is a total mismatch between the resources available and the task at hand, and it is quite unrealistic to expect stock assessments to virtually ever be done on coral reef fish [207]. There have been a few exceptions [211–213] (which assessed 25–35 species) however it appears that stock assessments of all species in a coral reef fishery have never been carried out at any location, ever. Fisheries models have also been used for the management of black corals and gorgonians, but only one or two species at a time [232,233]. The Schaefer model only requires catch and effort data to do a stock assessment, and is thus less expensive and laborious than other types of stock assessments. However, it assumes that Catch Per Unit Effort (CPUE) accurately reflects stocks, and CPUE has major flaws [198], considered in Section 3.2.

Most of the world's coral reefs are in low-income countries [234]. The global human population is growing rapidly, particularly in low-income tropical countries. Hundreds of millions of poor people live near coral reefs and depend on reef fishing for their livelihood and nutrition. The fishing pressure on most of the world's reefs is enormous, and there is a great need for information to manage reef fisheries so that the fish stocks are neither overfished nor are the people forced to go hungry. Most coral reefs have no information available to guide fisheries management [235]. However, a few developed countries do have coral reefs, and in fact Australia has the largest coral reef area of any

country in the world [54]. Stock assessments are not cost-effective for coral reef fish even in developed countries, but developed countries like Australia and the U.S. which have significant reef fisheries might be able to gather information on fish stocks in a way that was not as expensive as full stock assessments. We do not know the status of over 60% of the managed fish stocks in the U.S., most of which are not reef fish [236] and there are no stock assessments for 30 to >70% of major harvested fish stocks in developed countries [237], again because assessments are not cost-effective. Managing a stock without knowing its condition might be like driving with a windshield blacked out; crashes can be expected.

3.2. Catch per Unit Effort

An alternative method is often used, CPUE. In this method, catch is recorded and effort is recorded. If fish stocks decline, then the amount of catch may decline, but particularly the amount of catch per unit effort is likely to decline. Catch can be kept at previous levels when stocks decline by increasing effort, but CPUE would decline. There are a couple major problems with this. One is defining effort. Effort has to include the gear and methods, not just the number of person-hours worked. If gear and methods are changed so that efficiency increases, the ability to catch fish may increase. In many fisheries, changes in gear and methods occur all the time, and it is hard to judge how much they increase efficiency, but fishers adopt them precisely because they increase efficiency. If efficiency increases, CPUE can be kept high as stocks decrease, so CPUE fails to reveal decreasing stocks. In addition, fishers may shift from stocks that have been fished down to previously unfished stocks, maintaining high CPUE in spite of the serial depletion of stocks [61,62]. Thus, to get good CPUE data, it is often necessary to do scientific fishing, where the method is standardized and not changed over the years so there is no hidden increase in efficiency or effort.

Another major problem with CPUE is the problem of shifting baselines. A decrease in CPUE can indicate a decrease in stock, but it can not tell you the absolute value of the condition of the stock. If catch and effort were recorded from the time an unfished stock was first fished, then it could provide a realistic value for the condition of the stock, but that virtually never happens, and certainly not for coral reef fish. Assessment by CPUE is highly vulnerable to shifting baselines. A good example comes from Guam, where bumphead parrotfish schools were present in the 1960s, were spearfished out in the 1970s [238], and are now either locally extinct or very close to extinct [74,75,239] (G. Davis reports having seen a few recruits in recent years [240]). Data records on reef fish catch and effort go back farther in Guam than any other U.S. Pacific area, yet they only go back to the mid-1980s [241]. By then, these fish had already been fished out. CPUE also cannot alert the manager to the situation where a fish stock is in a highly overfished state, yet fishing continues which can catch few enough fish that the small catch is sustained over a long period. CPUE would record no decline in that situation.

A third and final flaw is that the amount of data available for any one species depends on the condition of the stock. Fish species which are overfished, economically and ecologically extinct and on the verge of local biological extinction, will have the least data in the fish catch (no data in extreme cases) and so any conclusions will be highly uncertain. Plus they cannot be easily distinguished from species that are naturally extremely rare. This problem extends to most of the fish catch, because of the many different types of gear used in coral reef fishes, such as spearfishing with and without

scuba, at night or day, traps, throw nets, gill nets, beach seines, rod and reel, bamboo pole, hand line, gleaning, *etc*. Each type of gear is a different kind of effort, with different sorts of selectivity, such as spearfishing being able to take herbivores while baited hooks generally do not. As a sampling technique, fishing has many biases in it, not under the control of the scientist, and many may be unknown. Every combination of gear type and fish species must be kept separate because they have different abilities to catch different fish. Add to that the fact that for recreational, subsistence, and artisanal fisheries, the provision of data by fishers is optional and data collection is almost always sampling a very small proportion of the catch (which itself is a small sample of the stock), and the result is that even for abundant species, there will not be enough data for analysis by species. One response to this is to lump together species into families to get enough data, but lumping is quite dangerous as explained above. CPUE has too many major flaws to be useful for coral reef fisheries management.

Catch itself is sometimes used to draw conclusions on stocks. Decreasing catches of reef fish could indicate that the actual stocks of live fish in the water were decreasing too [26]. That might work sometimes, but catch is a product of both stocks and fishing effort. If there is no fishing effort, nothing will be caught, but lack of catch does not indicate that the stocks themselves are at low levels. Increasing human populations strongly suggests increasing fishing effort, but again that is not necessarily the case. In American Samoa, population has increased rapidly, and long term decreases in catch were reported to show that stocks had declined [242]. Evidence from Underwater Visual Census (UVC) shows stocks have not declined, and creel survey data show that effort has declined [243]. Such changes have also occurred in parts of Fiji [184,244]. The key is that increased prosperity has led to a shift from fishing to store bought food [243]. Catch alone is not as good as CPUE for inferring stock levels, but either catch or CPUE can indicate a possible problem that should be investigated.

Catch has been used as an indicator of whether reef fish are being fished sustainably or not. The hypothesis is that high reef fish catch (expressed in weight per unit area of reef) increases the chance that the fish community is being fished unsustainably. In one study, low fish catches were considered evidence of either underexploited stocks or overfished stocks, with the decision of which based on literature and questioning local scientists and fisheries officers. Catches were acknowledged to be higher on actively growing reefs than sand, rock, and other substrates. Fishing effort was presumed to be proportional to population, based on published reports [26], but exceptions can occur as in the examples given above. In addition, all species were considered together, which is problematic for reasons given above. In a second study, a model based on a similar analysis of another data set predicted changes in fish body sizes that were likely due to fishing pressure. Catch could also reflect the local abundance due to natural factors such as habitat, or anthropogenic factors, so caution in interpretation was recommended [245].

3.3. Proportion of Stock Caught

A third alternative would be to measure the proportion of the stock caught per year. This would require both fish catch data and fish stock data. For most fisheries, this would require a stock assessment. However, coral reefs are relatively shallow, with clear water, and it is possible to count fish directly visually, as well as to estimate their size and from that calculate their biomass, something

not possible in other fisheries. In general, direct observation and counting of fish should be preferable to highly indirect methods such as stock assessment models. However, models can predict the maximum amount of sustainable fishing, which the direct observations cannot do. Direct counting is done routinely in monitoring programs on coral reefs, in UVC. Those data are already available for some locations. The U.S. Magnuson-Stevens Reauthorization Act of 2006 mandated that the regional fisheries councils of the U.S. set Annual Catch Limits (ACLs) for all U.S. fisheries by 2011, and the Western Regional Fisheries Management Council based in Honolulu decided to use this procedure for coral reef fisheries in the U.S. Pacific because they are "data poor" [246].
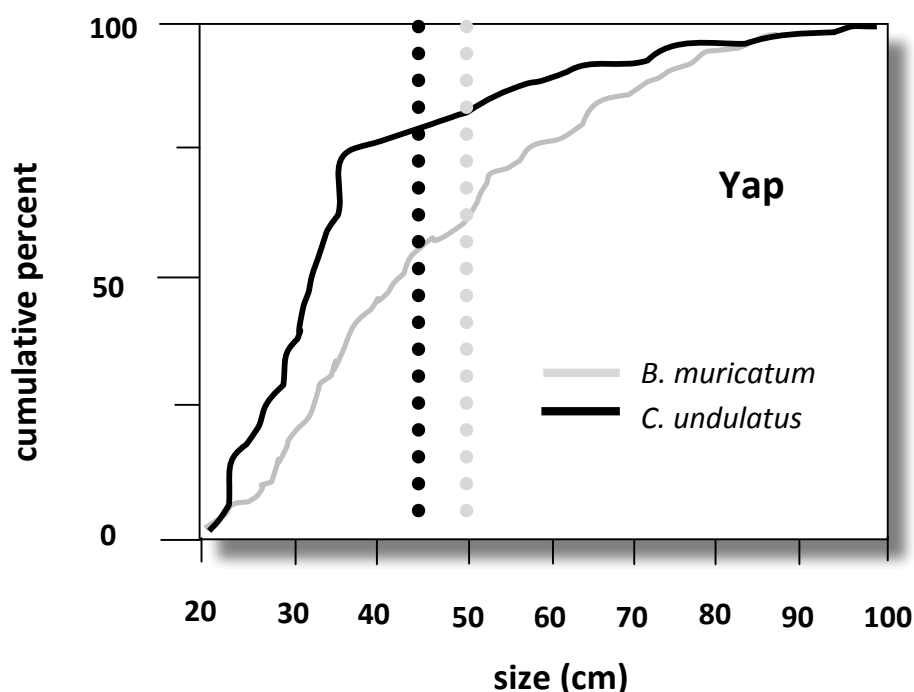
Monitoring programs were not designed to collect data for fish stock assessment, and may have a variety of shortcomings if used for that purpose, such as recording fish at only one depth, not getting a good sample of nocturnal or cryptic species (though most cryptic species are too small for fisheries), undercounting fish that strongly avoid divers [247–250], having a size-selective bias and species-specific differences in detectability [251] and over-counting fish that are attracted to the divers. When a UVC program is designed for stock assessment, some of these problems can be reduced [252]. For all the weaknesses of UVC, it is a direct measure of fish stocks while stock assessment is highly indirect, requires a variety of dubious assumptions, and is prohibitively expensive. As John Shepherd once said, "counting fish is like counting trees—except they are invisible and they keep moving" [201], but reef fish are unusual, they can be seen. In addition, recreational or subsistence catch data is a very small sample of a sample (catch) of the population, while UVC data is a direct sample of the population. But UVC requires training, scuba gear and often requires a boat. While probably not as expensive as stock assessment, it still is a significant expense, low-income countries do not have the financial resources to do this, and the coastlines of countries like Indonesia are far too long for this to be a practical alternative for them [235]. Because the method depends on fish catch data, it heavily biases the results to species that are in good condition, and is incapable of providing reliable results for species in the worst condition. In addition, this is a measure of fishing pressure, not stock status, so whether stocks are in an overfished condition or not is not revealed by this method. Knowing stock status is vital for knowing whether fishing pressure needs to be reduced or can be increased. Different species have quite different vulnerabilities and resiliences [209], and thus the reference point in the ratio that corresponds to fishing pressure at MSY would be different for each species, and could be widely different between species. Such reference points could be calculated, but appear not to be readily available.

3.4. Proportion of Catch Reproductive

A fourth alternative might be to record fish sizes in fish catches, by species, and determine what proportion of the stock is over the size at which the species becomes reproductive (Figure 4). This is relatively easy and inexpensive. Village fishers can usually identify many of the fishes in their catch, and could easily weigh and measure their catch, and the data can be gathered. One possible drawback is that the minimum reproductive size is needed. Minimum reproductive sizes can be calculated from data in the FishBase life history modeling tool for many reef food fish species, but not for each location [253]. A value like a mean of the available values could be used if variation between locations is not large; this is better than no evaluation of the species. A standard benchmark would be that at least 50% of the catch needs to be over minimum reproductive size. But if minimum reproductive sizes

are not available for species of interest, the expertise, time and expense needed to gather the information to determine that would be a significant hurdle. Another problem is that some reef fish grow quickly initially, but then stop growing and live long after that [254], rather like humans. If reproductive maturity is reached after growth stops, size will not be a good indicator of maturity. However, it seems likely that maturity will often occur at the time growth stops, as energy is diverted from growth to reproduction. Because it depends on fish catch data, the results will be most reliable for species in good condition and least for those in the worst condition. This method has been used in Micronesia, where for many species over half the catch was immature, and for some 100% are immature, leading to the conclusion that those fisheries are unsustainable [255,256]. The percentage of the catch that is under minimum reproductive size (L50) depends on not only the condition of the stock, but also selectivity of gear. It is most directly a measure of fishing pressure, but also indirectly of stock condition. Gear that is highly selective for only the largest individuals could produce a catch that would have half under L50 only for a heavily overfished species, and it could easily return false negatives. If a gear were selective for small individuals, the data could indicate recruitment overfishing occurring. If a gear is non-selective for size, the ratio might well accurately reflect the stock condition.

Figure 4. Cumulative size-at-capture graph for bumphead parrotfish (*B. muricatum*) and humphead wrasse (*C. undulatus*) in Yap, redrawn from [256]. The vertical dotted lines are the L50 for each species, the median size of first reproduction. Both species were harvested below their L50, *C. undulatus* more than *B. muricatum*.

3.5. Trophic Level

A fifth alternative might be to measure the mean trophic level of a reef fish community. This is because fishing typically removes large predatory fish first, then the next smaller fish and so on down the food chain. This is called "fishing down the food web" [109]. A measure of the mean trophic level

of a reef fish community would provide a measure of how far that process had gone, if the mean trophic level were available for near-pristine reefs to compare with. Given the great loss of apex predators (Section 2.3), most reefs should show significantly lower mean trophic level. However, that would not provide information on individual fish species, nor would it indicate whether species were overfished. In addition, there would be the expense of gathering and analyzing this information.

3.6. Virgin Stocks

A sixth alternative would be to gather information on reef fish stocks and communities on unfished control reefs. If UVC data were collected on such reefs, then the stock level below which the stock is overfished could be determined, since fisheries models indicate that level (MSY) is at 1/2 or 1/3 of the unfished biomass (depending on the model). Note that the fact that a stock is below the biomass of a pristine reef does not indicate that it is overfished, it must be below about 1/2 to 1/3 of that of a pristine reef. UVC data could be collected at the site of interest, and the stock condition for each species determined. In recent years, UVC data has been collected on a variety of remote near-pristine coral reefs in the Pacific [16,23,29,32,105–108], so for the first time, such data are potentially available. But there are a variety of significant problems with this. For one, the remote reefs will not be the same as any reef you wish to evaluate stocks at. The remote islands are always smaller, because they must be too small for people to live on or else there would be people living there catching fish. Island sizes can cause a wide variety of differences; high islands have more runoff and thus probably have higher nutrient levels in reef waters and more plankton and productivity. High islands may only have fringing reefs, while tiny islands may be on atolls with lagoons and have habitats that high islands do not have. They may be at different latitudes, such as the NW Hawaiian Islands which are high latitude reefs, with colder water and fish communities that are different in some ways. The remote islands are likely to be long distances from the islands where you wish to study stocks, increasing the chances of biogeographical differences in fish assemblages, differences in oceanographic conditions, *etc.* A study that looked at the importance of fishing and habitat on fish populations in Fiji found that decreases of fishing pressures in some areas led to increases in picivores and species in the fishery, but picivore populations were more strongly influenced by the reduction in prey due to the loss of *Acropora* habitat. The effects of fishing and habitat on individual species were stronger than on functional groups [184]. A study that correlated reef fish abundance and biomass with fishing and habitat at 63 sites in 17 countries and territories across the Pacific found that 20% of the variation was correlated with fishing and 30% was correlated with habitat [257]. There is considerable variation in near-pristine reefs, with many being atolls with lagoons but the northern end of the Marianas Islands being high islands without lagoons. Further, near-pristine reefs have been found in a wide variety of latitudes from the equator to some of the highest-latitude reefs known, and in widespread areas of the oceans with different oceanic productivity. If a group like apex predators is almost always in high abundance on near-pristine reefs and low abundance near people that should be enough under the precautionary principle to justify action. Some large fish like giant grouper (*Epinephelus lanceolatus*) appear to be rare everywhere [258,259], so their rarity near people is unlikely to be caused by fishing. A variation on this technique is to correlate abundances of a species with fishing pressure. Humphead wrasse (*C. undulatus*) and bumphead parrotfish (*B. muricatum*) abundances have been shown to be correlated
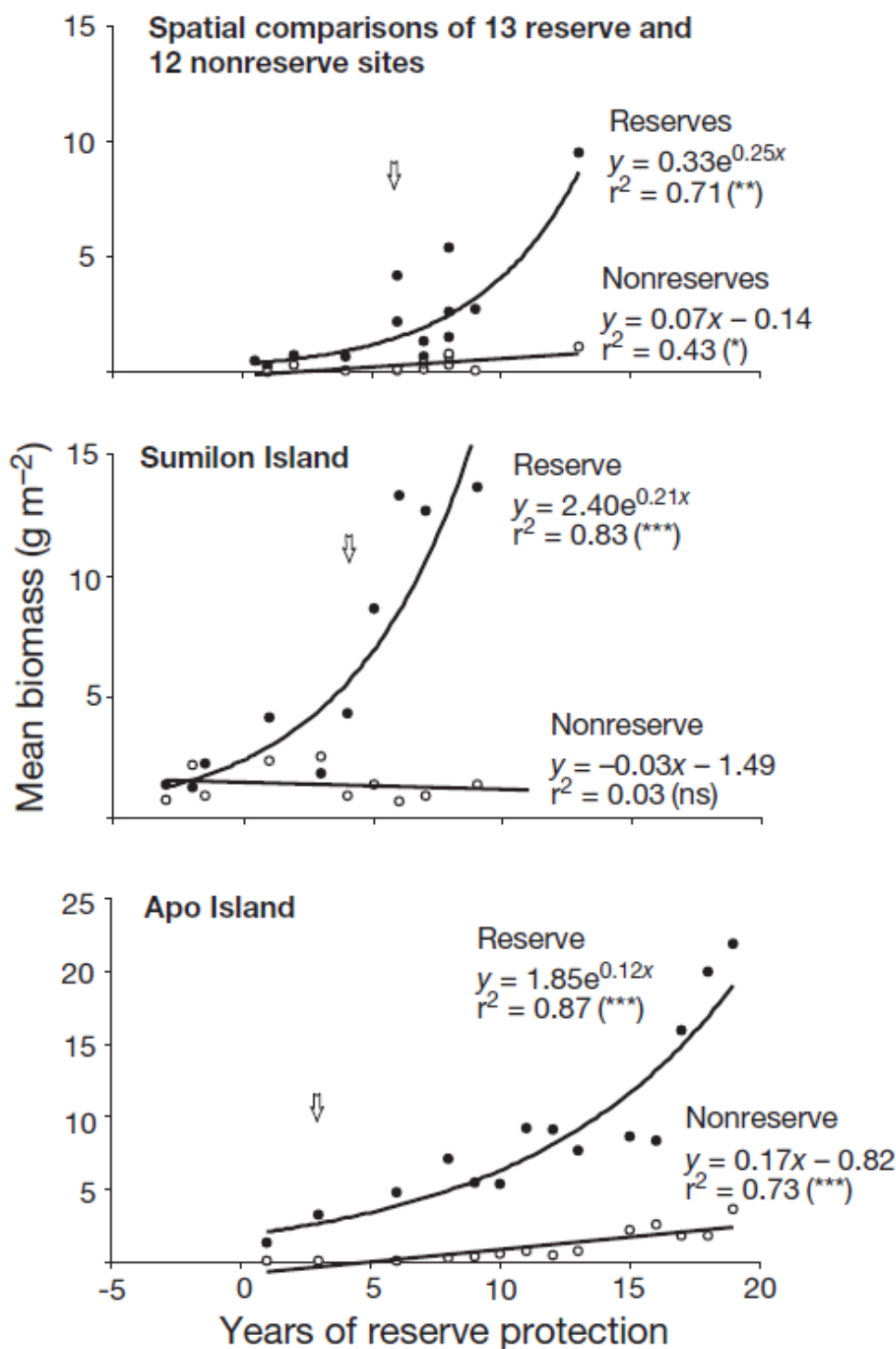
with fishing pressure [111] and human population [110], respectively. Human population may be a proxy for fishing pressure, but other things correlate with human population as well. Abundance of species at near-pristine reefs can help at fished locations to distinguish naturally rare fish from fish that are rare due to fishing.

In addition, there are many activities other than fishing which people do that affect reefs and thus indirectly affect fish populations, by producing sedimentation, nutrient and pollutant runoff, *etc.* Some of these differences can be controlled for by comparing small fish as a group separate from large fish. The small fish can serve as a control group for many of these factors, while the large fish will be differentially affected by fishing. If large fish have lower biomass near people than at remote reefs while small fish have equal biomass at the two, the effects on the large fish are hard to explain by anything other than fishing [107]. Indirect effects such as sedimentation and nutrients must have demonstrable effects on the reef habitat on the inhabited islands to be viable explanations of differences in fish populations. This method is already being used in the "Coral Health Index" [108], there is a baseline for herbivorous fishes as a whole [107], and it would be very useful for monitoring functional groups of herbivores [169]. However, while this method may be useful for trophic, functional, or size groups of species, it may be less helpful with individual species. If a species like grey reef sharks (*Carcharhinus amblyrhynchos*) is abundant on nearly every near-pristine reef, the inference may be fairly strong that the natural abundance of them is high. Because this method does not rely on fish catches, results are just as reliable for groups or species in poor condition as those in good condition.

3.7. NTA Control

A closely related seventh method would be to use an NTA as a control area, and calculate a ratio of populations outside the NTA compared to inside the NTA [260]. That ratio would be an indicator of the impact of fishing, and it would only require the presence of an NTA and data from inside and outside in the form of UVC or standardized scientific fishing. It would be less subject to differences between very different reefs and islands that are far away and may be in different oceanic environments. It would provide a better control for local conditions and temporal variations in recruitment. The major drawback is that it assumes that fish stocks within the NTA have returned to close to that of an unfished reef, and doesn't work well for fish that move over large areas [260]. For coral reefs, the evidence indicates that medium size predators within NTAs continue to increase for decades ([261,262], Figure 5), and NTAs do not have abundances of the largest fish species such as sharks anywhere close to that of a natural unfished reef [23,260]. Most NTAs on coral reefs are far too small and have been established for far too short a period to have shark populations anywhere near that of unfished reefs, where they compose about half the fish biomass. One exception is large NTSs (over 100 km^2) in the western Caribbean which have been well enforced for over 10 years. For some of them, biomass and apex predators can approach that of near-pristine reefs in the Pacific [130], though it is possible that those NTS sites were chosen in part because the fish populations were exceptional before protection (due to either lower fishing pressure or other factors). NTA controls might work well for small fish, but then they are the least likely to be overfished.

Figure 5. Strong evidence of increases in biomass of "large" predatory reef fish (snappers, medium groupers, *etc.*) biomass inside reserves greater than in non-reserve control areas in the Philippines. The top graph shows several sites established for different durations with data taken at the same time, and the bottom graphs show two sites over time. Reproduced from [262] with permission.

3.8. Large Fish Abundances

Another indicator is the abundances of large fish species, since they are likely to be most affected by fishing. This would probably best be done with UVC, which if there are monitoring data available,

is relatively easy. If monitoring data are not available, this method would also require a substantial expense. This might produce useful information about the overall condition of the reef fish community, particularly when coupled with information about what fish communities are like at near-pristine reefs. About half of the total biomass is in apex predators at near-pristine reefs, so if the reef of interest has much lower levels, fishing appears to have had a substantial impact on that part of the community. Without some way of knowing what the natural population of a particular species would be on that reef, it is hard to know more about its condition. A variation on this method could be a rule of thumb, such that the amount of fishing would be indicated by the largest fish that are abundant [263]. A reef dominated by apex predators would be considered pristine, a reef where most of the remaining fish were damsels or smaller would be surmised to be almost completely depleted by very heavy fishing. Size distributions for the whole fish community tend to be much more stable than for individual species [264] and it would take much less effort to get a large enough sample than for species. Graham *et al.* [265] propose a more quantitative way of doing this. For individual species, the size distribution could provide information on the condition of the species, so a species with very few large individuals may be overfished. But it would be necessary to know what the size distribution was for an unfished stock, because large individuals are naturally less common than are small individuals. Such data probably exists now for many species. The assessment would depend on having a large data set on the species, since the largest individuals are the upper tail of the distribution and thus much less reliable than measures of central tendency. Another variation would use the median size and thus would not require large data sets. This might not work well with species that stop growing [254]. A variety of quantitative size-based indicators can be helpful in assessing reef fish stocks, including the size spectrum. The slope of the size spectrum can be an indicator of fishing pressure [251]. Changes in the slope, like changes in abundance and/or biomass, can indicate whether a fish community or stock is increasing or declining. However, it doesn't reveal the condition of the stock, whether it is overfished or not. Using changes over time is subject to shifting baselines, like CPUE. It also requires UVC data which is too expensive for poor countries.

4. Fisheries Management Tools

4.1. Effort and/or Catch

In developed countries, many tools have been created for controlling fishing. Common tools are effort controls such as fishing seasons and gear limitations, and catch limits such as daily bag limits, size limits, and annual catch limits. Most such tools are rarely used in developing countries, where most coral reefs are. In most developing countries, compliance with laws adopted by central governments is low [266,267]. The government does not have the money to hire enough enforcement agents, knowledge of the law by citizens is likely limited, and citizens circumvent many laws. (Unpopular laws are very hard to enforce in any democracy, whatever the development status is.) In most countries with coral reefs, people must obtain fish to feed their families. Effort and catch are very hard to control in low-income countries [268]. Minimum size limits would be most enforceable for commercial catches, since the catch would be on public display, but most reef catch is surely subsistence. However, gear restrictions, such as banning destructive fishing such as blast and poison

fishing, can sometimes be effective, depending on enforcement capabilities [155]. Fishers may actually prefer gear restrictions, since they often have several types of gear and can switch to different gear and continue to fish. Different types of gear take different proportions of the fish such as herbivores that are thought to be needed for resilience. Spear guns and traps take a higher proportion of the species needed for resilience, while hook and line takes a lower proportion. Beach seines are the most destructive to habitat and take juveniles. Managing with gear regulations requires tradeoffs between minimizing gear conflict, reducing overfishing, maintaining profits, and protecting species affected by coral losses and important for resilience [192]. A second problem comes from unselective fishing gear and the need to manage each species with different regulations, and the very large number of species in the fishery. Some types of gear such as hook and line, gill nets or traps are not very selective. If the gear is unselective, then fishers cannot avoid catching particular species, and may not be able to avoid catching fish that are too large or small. If, however, the fish are still alive when removed from the water, they can be released alive, though mortality may still result for some (mean 18%, median 11% [269]). Spearfishing is potentially highly selective and so the fisher can choose to avoid any species or size they choose, though size has to be estimated in a short period of time. However, in practice spearfishers may spear anything they see, making it very unselective [270]. In any case, it would appear that there would need to be a myriad of regulations [271], because each species would have to have different regulations, such as different size limits since all are different sizes, or catch limits since the condition of each stock will be different. The catch is mainly concentrated in just a few species or species groups, and that might be used to reduce the number of regulations [255,256], but that would probably miss most of the species that are overfished. Most of the commercial catch is often from spearfishing [255], and spearfishing has the potential to be very selective. Banning night spearfishing or SCUBA spearfishing is a significant control on effort. A SCUBA spearfishing ban is accepted and appears to have relatively high compliance in American Samoa [272], but a night spearfishing ban would be difficult to implement in Pohnpei [255].

Minimum mesh size regulations for nets and traps can effectively be minimum size regulations for those types of gear. Where traps or nets are the dominant gear used (as are traps in the Caribbean) this may be an effective way to institute minimum size limits. However if they are a minor component, it may have little effect. Traps also can be very powerful fishing tools, since they fish even when the fisher sleeps, and if they become lost they can continue catching and killing fish long after the catch is no longer collected by the fisher. This is a particularly wasteful and destructive result, which can be countered by requiring traps to be made out of degradable materials. That increases the cost to fishers for repair and replacement of traps, but the cost can be reduced by only requiring degradable escape hatches.

4.2. Incentives, Rights-based Management and Individual Transferable Quotas

Some fisheries scientists (e.g., [198,273]) point out that the incentives in most fisheries are to catch the fish before someone else does, a "tragedy of the commons" situation. This leads to the "race for fish." The argument is made that if the incentives were changed, then the behavior of fishers would change because people respond to incentives [273,274], which has empirical support [274,275]. Rights based management advocates argue they promote conservation, capacity reduction, improved product

quality, and maximizing economic efficiency [276]. One way to try to do that is ITQs or Individual Transferable Quotas, one of many types of rights-based fisheries systems. Each fisher would own the right to catch a certain portion of the total catch that was allowed. There would be no incentive to rush to catch the fish first. Individuals could buy or sell ITQs to increase or decrease their catch. ITQs have been reported to have positive effects on ecosystems [277]. Critics have pointed out a variety of problems with ITQs [278], including leading to negative socio-economic and cultural impacts, increased management costs, encouraging concentration of fishing quotas, and producing resource damage [276]. Others say that setting up ITQs is privatizing a public good [279–281] and that rights-based systems have socio-economic shortcomings that trouble many [282], though ITQs could be auctioned each year much as rights to utilize publicly owned timber or rangeland are. ITQs worked for trochus on an atoll in the Cook Islands [283]. The problem for most reef fisheries is that the large number of fishers spread out along a long coast could make administering this system nearly impossible.

4.3. Ecosystem Approach

An ecosystem approach to fisheries (EAF), ecosystem approach to fisheries management (EAFM), ecosystem-based fisheries management (EBFM), ecosystem-based management (EBM), or ecosystem management (EM) is usually viewed as managing the effects of fisheries on ecosystems [181]. It is not incompatible with managing individual fishery species. No one seriously proposes abandoning single species management [281], and often it is thought of as adding ecosystem considerations to single species management; both are needed and interdisciplinary work is needed [284]. Some say the goal is sustaining healthy fish stocks and healthy ecosystems [285] while others say that ecosystems need to be rebuilt [281]. Another view is that it means managing for present benefits while preserving the option of future generations deriving benefits [286]. EAF requires sustainability in ecological, economic, and social spheres [287,288]. EAF recognizes that fishing affects the ecosystem, such as by damaging habitat or producing bycatch mortality or by producing trophic cascades or phase shifts, and that damaged ecosystems reduce the ability of the ecosystem to support fish stocks and fish catches. Focusing only on catches of the target species can be short-sighted, if ecosystem damage ends up reducing fish catches. Fisheries can cause major changes in communities of fish and other species, reduce diversity, remove keystone species thus causing major ecosystem changes, and cause the loss of functional groups and thus function [72]. Destructive fishing such as blast fishing [289] and poison fishing (such as cyanide used to catch aquarium fish [290]) are common on coral reefs in some areas, damaging habitat and reducing coral diversity [291]. However, bycatch is not a problem on coral reefs. All fisheries are multi-species, most with target and bycatch/discard species, while coral reef fisheries are very efficient and all of the many species are kept and eaten [224] except for a few poisonous species in some cases. "Target" may be a less appropriate term with coral reef fisheries than just "catch," since target implies fisher choice of species, while most reef gear is non-selective and even in spear fishing (which is potentially the most selective), fishers may take fish primarily opportunistically, in practice very non-selectively [270], and all catch is eaten. There are exceptions, such as fisheries that target a specific species such as bumphead parrots [74] or humphead wrasse (in a Guam spear fishery [292]) or a limited group of fish as in the live food fish trade [224], or a variety of reef invertebrate fisheries (e.g., sea cucumbers, [62,293]). A broader view of EAF or EBFM would be

managing the ecosystem as a whole, including fisheries. This is particularly important for coral reefs, where other threats such as sediment, nutrients, diseases, bleaching and acidification can have major impacts and reduce fish catch by damaging habitat, as well as reduce other ecosystem services. The impact of reef fisheries on resilience and the loss of corals which degrades fish habitat, is an example of such wider concerns. Understanding and managing ecosystem health [294], ecosystem integrity, or state [284] are important along with maintaining ecosystem services, and there are many ecosystem services in addition to fishing. Indicators should guide management to avoid unsustainable impacts of fishing on ecosystems [286]. Emergent properties of ecosystems could be used as indicators or metrics [295] of ecosystem health [263]. Productivity, diversity, and resistance (defined as ability to resist a phase shift) [294] have been proposed for ecosystems in general, but those might be hard to measure on reefs on a routine basis except for fish diversity, and the standard for that would depend on the biogeographic location. Specific proposals for indicators for coral reef health include the biomass of large predatory fish relative to that on near-pristine reefs, biomass of herbivorous fish, density of microbes [108], different functional groups of herbivores [169], high diversity [289,296], high total fish biomass, low disease prevalence, and ability to calcify faster than erosion [296]. It is likely that we cannot determine the health of the ecosystem by monitoring any single species [297]. Natural history is necessary as well as quantitative ecology to understand ecosystems [298]. Models can be used to explore the effects of fisheries on the broader ecosystem and vice versa. For example, a model predicts that only small amounts of fishing in Kenya can cause urchin populations to suddenly increase [267]. The same model predicted that fishing only piscivores minimized ecosystem impact but provides low catches, while fishing herbivores and piscivores increased catch and impacts. MPAs have been proposed to be a central tool in EBM [299]. Ecosystem plans, however, have often omitted many of the basic principles of EBM, such as being based on science, and including monitoring, adaptive management, inclusion of humans, sustainability, and ecosystem health [288].

EBM may provide indicators of the status of stocks, such as the virgin stock controls on near-pristine reefs (Section 3.5). Although such indicators could easily be used in "dataless management" (Section 4.6), managers in developed countries have little precedent for how to integrate it into management decisions which conventionally would be based on stock assessment models or CPUE. Ways of integrating it need to be developed, especially since stock assessments will virtually never ever be available for coral reef fisheries. Distrust of ecological indicators may initially be widespread among conventional fisheries managers in developed countries, even though the need for EBM is widely acknowledged.

4.4. Alternative Income

Economists have pointed out that, in developing countries, most fishers have very limited alternatives for their livelihood. There are no other sources of employment that pay better. Desperately hungry people will take the last fish they can get [231]. If better paying alternative livelihoods were made available, people would move from fishing into those other jobs [300–302]. Poorer fishers are less likely to leave a declining fishery due to "poverty traps," and fishermen that live in households that have more alternative income sources are more likely to leave [302]. But providing better paying jobs for millions of people in rural areas of low-income countries is a near impossible task. Aid agencies

and government agencies have been working to do that for decades, with limited success. Improvements in economic conditions are primarily driven by macroeconomic changes, not economic aid, and these changes often happen first in cities. However, in American Samoa [243] and parts of Fiji [184,244], increased income has led to more purchase of packaged food and less fishing. In the Indian Ocean, reef fish biomass decreased strongly with economic development to a minimum point, and then increased with further development. There was only a weak correlation with population, and a moderate correlation with fish habitat. Economic development can affect the environment in nonlinear and sometimes positive ways. The U-shaped function is consistent with the Kuznets curve hypothesis, that development first causes more damage to the environment but then less damage with increasing affluence as people want a better environment [303]. Though an increase in better paying jobs sometimes happens independent of management, this is often not feasible as a way for managers to reduce fishing pressure on reefs in low-income countries.

4.5. Spatial Closures: MPAs and NTAs

Conventional fisheries management tools in developed countries include spatial closures. Such closures are usually temporary and removed once stocks have recovered. MPAs have received a great deal of attention in recent years. MPAs provide some level of protection for a specific area for an indefinite period. An NTA, no-take marine reserve (NTMR) or fully protected marine reserve is a type of MPA where all extractive activities are forbidden, while a Marine Managed Area (MMA) is an area where some extractive activities may be forbidden but others allowed. Many other names have been used as well [304]. A no-go area is one in which entry is not allowed; no-go areas on the Great Barrier Reef had higher shark populations than NTAs [23]. In the western Caribbean, the highest fish biomass and apex predator populations are only in the largest MPAs (100 km^2 or more) which have been well protected for a decade or more [130]. MPAs can have areas within them zoned for different types of use or protection, such as the Great Barrier Reef Marine Park [305]. MPAs have many objectives and potential benefits, primarily in the areas of conservation and fisheries [305–310]. Research has generally shown NTAs to be more effective than areas where some extraction is allowed [305,309] and have been shown in over 50 cases to have greater numbers and/or sizes of exploited species or other benefits within their boundaries [71,268,302,311–314]. A review of studies reported average increases of fish biomass in MPAs of 413% (N = 56 studies), a 200% increase in density (N = 31), an 82% increase in size (N = 20) and a 71% increase in diversity (N = 32) [308]. Some studies have reported large increases in abundance and biomass over time (e.g., [176,261,262,315]) (Figure 5), but others have not [71,268,316]. One study found higher levels only in fished species, only in piscivores and herbivores, with increases over time significant only in piscivores [317], patterns supporting the view it was due to protection. In another there were higher abundances inside than outside, with the largest effects with the largest species and the effect decreasing with size until there was no effect with small species [318]. A review of MPAs in the Philippines concluded that a majority of MPAs improved fisheries, biodiversity and livelihoods, and that community-based MPAs had fewer conflicts than government-run MPAs [319]. One study found that compliance was the strongest variable affecting MPA effectiveness [320], which makes sense because compliance reduces fishing pressure. Another found that the fishable biomass in the MPA increases as fishing restrictions become more stringent [205].

MPAs are also predicted to produce "spillover" of fish into nearby open areas. The thought is that increased abundances of fish within the boundaries will lead to competition for limited resources, and some fish will move outside the boundaries where abundances are lower and competition less. Although there is evidence of spillover, it has been more difficult to demonstrate than increases inside the MPA. Among other things, demonstrating a net spillover effect requires finding that migration out of the MPA is greater than migration into the MPA [268]. An analysis of the literature on spillover reported that it was found in 14 of 16 studies, in 30 of 35 taxa [321]. Spillover can be indicated by fishermen "fishing the line" just outside the boarder of the MPA where spillover would occur. A third predicted effect is the "recruitment effect," "egg subsidy," or "larval export." MPAs are predicted to allow individual fish to grow larger than outside the boundaries, where they are taken by fishers. Fecundity increases much faster than length, such that a larger fish produces orders of magnitude more eggs than a smaller fish, plus the eggs and larvae are larger for some long-lived species [322,323]. Thus, the larger fish in MPAs are predicted to increase recruitment over an area mostly outside the MPA (because of egg and larval dispersal in currents). Several studies have documented increases in egg production in MPAs, a study reported higher larvae densities inside an MPA, and another study reported higher settlement in a protected area [324], and there are several other studies that have documented egg subsidy [312]. Most recently, genetics has been used to find fish far outside MPAs whose parents were found inside the MPAs [325]. Large fish may also serve ecosystem functions which small fish are unable to, such as eating more algae [128,323]. A modeling study concluded that for species that are site-attached and thus there is no spillover, NTAs improve on the performance of conventional fisheries management only if undersized individuals are taken in the fishery [326].

Although the relative sizes of spillover and egg subsidy effects are not known, it seems possible that the egg subsidy effect may be much larger than the spillover effect. Recommendations have been made that a minimum of 20% of coral reefs should be in NTAs, and that there should be networks of MPAs so larvae from one supports another. The percentage of reef areas in NTAs may best be adjusted to fit the purpose and local situation [304]. It is commonly said that MPAs must be complimented by good fisheries management outside the MPAs, however, what that good management would be is not often specified (perhaps because that is far more difficult). MPAs have been reported to have less disease [102], fewer crown-of-thorns starfish outbreaks [98], and less decline of coral cover [103,104]. Also, fishing selectively removes the larger individuals, and causes the species to evolve into small, early-maturing fish. Modeling indicates that MPAs can stop or reverse that trend [327].

It has also been pointed out that MPAs do not protect against everything. They protect against overfishing, but they cannot protect from sea temperature rises causing mass bleaching and coral death [165,168,189] and in some cases from ensuing phase shift to macroalgae [168], nor from ocean acidification or introduced species [328] nor hurricanes and oil spills [329]. Unless the protected area is coupled with management on land ("ridge to reef"), it cannot protect against sediment, nutrient, or other pollution runoff [330]. If MPAs are opened to fishing periodically, the largest fish are usually removed, greatly reducing their ability to provide egg subsidy. Some MPAs have been declared by central governments, without any local support, and without the ability to enforce them [331]. Fishers and local residents then often ignore them, and they are called "paper parks" since they exist only on paper [265,331]. Top-down, command and control policies can lead to feelings that the MPA is illegitimate, leading to conflict and low compliance [332]. Ardent opposition from fishers, particularly

in developed countries, and the need for intensive work with communities, are drawbacks for MPAs. Many MPAs are not enforced adequately or are ineffective [11,54,104,333], with one review of 1,300 MPA's with subtidal components reporting that 9% have effective management but 71% have "unknown management" [331,334]. Failure rates for MPAs are high. MPAs are not enough [37], they compliment other forms of fisheries management instead of replacing them [335], and they are no panacea [308]. MPAs can allow for a full age and size range in species, and thus provide a counterweight to the evolutionary selection for early maturity and small size that fishing imposes [323], they can keep stocks above recruitment limitation levels, and provide insurance against failures of other controls outside the MPAs [335,336]. MPAs can provide refuges that replace the refuges that formerly were provided by places fishing technology could not reach, but now can [281], such as deeper water [337]. MPAs can produce the same yields as effort controls under a broad range of conditions, but offer more sustainability [338]. Another study found that MPAs provided greater long-term catches with fewer restrictions than either minimum size limits or temporary complete closures [236]. One study argued that moving closed areas around could increase benefits. A model projected that stocks depended on whether fish in newly closed areas increased faster than they decreased in newly opened areas. It assumed that fishing pressure was evenly spread through the open areas [339], but usually fishers will heavily fish any newly opened area that has built up fish populations [268], and buildup in a newly closed area is slow [261]. It seems likely that in most situations mobile MPAs would not produce greater buildup than stationary MPAs. MPAs are particularly good for ecosystem-based management, since they protect everything, including things we don't yet understand [271]. MPAs are one of the best tools we have for coral reef conservation.

One advantage of MPAs is said to be that they require very little information, *i.e.*, no information on fish stocks is needed. But information on the habitat could help locate MPAs in areas with better than average habitat and thus better than average chances of success. Knowledge of current patterns could increase the chance of a network of MPAs supporting recruitment in each other. Knowledge of the home range sizes of different fish could help plan MPAs that were of the best size to maximize benefits [340]. For fisheries benefits, at least some fish stocks need to be in an overfished state, so some knowledge of the condition of fish stocks is important. So while no information is absolutely necessary, information may increase the chance of success, if the information can influence the location, size and spacing of MPAs. MPAs are also said to be less expensive [318] and also easier to enforce that conventional regulations [318], since fishers with gear are prohibited from the area.

There have been many criticisms of MPAs (e.g., [304,341]) and criticism of research on MPAs [268,342]. One criticism is that closing an area simply shifts fishing effort out of the closed area into open areas, and further depletes the open areas and can also expose fishers to more dangerous conditions for fishing [342]. At the same time the argument is often made that excluding fishing effort from the MPAs means that the closure causes a decrease in fish catch, so spillover must more than compensate for that decrease in fish catch [18]. However, if fish stocks outside of a newly closed area are no lower than inside the area, moving fishing effort outside the MPA should not necessarily lead to any decrease in fish catch. With time, if stocks outside the closed area are depleted more by the increased effort, total catch could decline. But most coral reef MPAs are so small and separated by such great distances, fishing pressure outside MPAs may be increased so little as to make detecting a difference difficult. Closing off small areas may not have much effect on total fish catch, the effect on

catch increases with increasing the percent of reefs in MPAs [336]. If the fish species outside the MPA are not fully exploited, catch may change little, but if fish are overfished there may be some decrease in catch; some suggest that concerns about displacement appear to be exaggerated [336]. Spillover did not compensate for lost catch in Kenya [343], but it did at Apo Island in the Philippines [344] which suggests small MPAs may have a better chance of producing net increases in catch. One model study has found that MPAs provide greater long-term catches while being less restrictive than either minimum size limits or temporary complete closures [338]. Other model studies differ on whether MPAs can produce net benefits [341]. It appears that the question of whether MPAs can, or frequently will, produce a net total increase in catch on coral reefs remains open.

Perhaps an unstated criticism of MPAs is that a closed area in a multispecies fishery is likely to both protect species which are overfished and need reduced fishing pressure, but also protect species that are not overfished and do not need protection. For fisheries, it is, in a sense, a blunt instrument, which is being used because there isn't a better instrument available. However, even if a species is not overfished by fisheries standards (that is, its biomass is not less than that at MSY or OY), the reduced stock may increase the risk that the reef has reduced resilience. Such is clearly the case with herbivorous fish which keep the ecosystem from shifting from coral to algae, or could shift it back. The fact that we do not know all the functions of all the fish, suggests that there is a risk of reduced resilience from decreased abundances of any species and the precautionary approach is to protect the resource until further information is available [336]. Further, if information is not available showing which species are in need of protection and which not, selective protection is not possible. Another unrecognized weakness is that MPAs (like conventional stock assessment) assume that unfished stocks will recover because they are density-dependent, an assumption that is not always valid [58].

MPA research has been criticized for weak evidence. There are more theoretical studies and reviews than empirical studies, reviews often cite other reviews, and many models are not checked against reality [342]. Objectives and assumptions that seem reasonable have become accepted as fact [342]. It has been said that supporters of MPAs have accepted that they enhance fisheries based more on faith than on evidence [345]. Most MPA studies have weak designs, and those with good controls are relatively rare [268,342] in part because they are difficult to do. In many cases, conclusions are based on higher abundances inside MPAs than in control areas outside, but some evidence (e.g., [316]) based on inside vs. outside may be due to areas included in MPAs being selected for better than average coral and/or fish [346]. Lack of increases over time would suggest that an MPA is not having an effect, perhaps because of ineffective protection, small size, *etc.* Increases over time inside the MPA without increases outside the MPA (as in Figure 5) are strong evidence, much stronger than just inside higher than outside, at one time or at all times. The evidence for spillover and egg subsidy, while there, has been thin until recently. The egg subsidy effect has been difficult to demonstrate, because finding ways of following eggs and larvae as they disperse have been near impossible, but innovative new ways such as genetics and isotope labeling are now being used to trace fish larvae [325], and effects on immobile species outside of MPAs have shown the effect [312]. For the spillover effect, if it was a large effect, it would be easy to demonstrate. But it has not been easy to demonstrate, in part because it may not be a large effect. MPAs are not favored because they are such wonderful fisheries management tools, they are favored because of the lack of viable alternative fisheries management tools for coral reefs [268], and because they are good conservation tools. It is

easy to say that effort and catch controls are better for some temperate or pelagic fisheries, but for coral reef fisheries they are usually not viable options, effort and catch are almost impossible to control in low-income countries. There are few if any coral reef fisheries that have been managed well using effort and catch controls [22].

Criticizing MPAs for not being good fisheries management tools is criticizing them for not doing well something that is not their primary function. The primary function of MPAs is conservation, protecting the resource in a time when most of the coral reefs of the world have fish species that are overfished, and overfishing along with destructive fishing, is the greatest local threat coral reefs face worldwide [54]. Even if MPAs are not great fisheries management tools, they are one of the best tools for coral reef conservation. They are one of the best tools for trying to restore herbivore abundances to the point that coral reef resilience is strengthened enough that reefs do not make phase shifts to macroalgae beds when bleaching kills most of the corals in the future. At least one study has reported that NTAs led to increases in herbivorous fish faster than carnivorous fish, and to decreases in algae over time [176]. Another found that an NTA increased large parrotfish and predatory groupers which in turn reduced smaller parrotfish, but the increase in large parrotfish led to decreased algae [128]. Further, it has not really been demonstrated that MPAs are poor fisheries management tools on coral reefs. Egg subsidy may be a fairly large effect, we don't really know how large yet. The recommendation of a minimum of 20% of coral reefs being in NTAs is based on calculations in which NTAs were treated as a spatial tool for making sure that there was sufficient reproductive stock to avoid the whole system from being overfished or worse [347]. If they work as predicted, they could be a good fisheries tool in that situation, which is probably a quite realistic situation for most of the world's coral reefs.

The reason that fisheries enhancement effects of MPAs have been treated as real when the evidence was not abundant is that the reality is that there must be local community support to establish MPAs [268]. MPAs are most needed where fishing pressure is heavy, yet in developing countries an NTA closes off an area to poor fishers that have a vital need to catch fish to feed their families. Local community fishers are very unlikely to support an NTA unless they realize some benefit from it, and depriving poor fishers of their food source is harsh treatment. Because national governments in many low-income countries are unable to enforce local laws without community support [266], community support is vital for the success of an NTA. If village fishers support the NTA, they can provide the enforcement which national governments are unable to provide. In the Philippines, the best way to convince villagers that an NTA is in their best interest is to take the fishermen to Apo Island, the site of the oldest continuously operating NTA, and have them talk to the fishers there. The fishers there are convinced that they catch more fish more easily, and the visitors usually leave wanting to try it in their village. Because they have such an urgent need for food to feed their families, poor reef fishermen will agree to most anything that will increase their fish catch. But we need to keep the reality in mind that the improvement is likely to be fairly small, and that with the rapidly growing population, the enhanced fishery will soon be unable to provide enough food for all the new people. As a fisheries tool, at best NTAs are a temporary solution to the food shortage problem, and if population growth rates are not reduced, the improvement will soon be lost. But the critical need to have local support means that MPAs are almost always sited where a community is willing to try it, not where science indicates it will be most successful [348]. MPA acceptance and thus success is likely to depend on social factors [331,349,350], particularly the most important goal for the society. If the most important

goal is food for very poor fishermen, MPAs are likely to be successful if they improve catches. If food for occasional feasts is the most important goal, then brief openings of more temporary closures may be preferred over permanently closed MPAs.

Because community support is critical to the success of MPAs, most MPAs are much smaller than they need to be, simply because communities are small and it is not possible to establish larger MPAs [348]. Hilborn *et al.* [351] give guidelines of when MPAs are most likely to be successful, and coral reefs fit most of their criteria for MPA success. One of the criteria is that the fish be relatively sedentary [312]. In particular, the fish should not have home ranges larger than the MPA. Most coral reef fish are relatively sedentary, so even small MPAs, like on Apo Island in the Philippines, can have demonstrable benefits. Even MPAs that are small and are based more on opportunity than science, can accomplish much [348,352]. However, the largest reef fish species, particularly the sharks, have large home ranges. In the case of sharks, home ranges of a few kilometers, or more is not unusual. About half of the entire coral reef fish biomass on near-pristine coral reefs is in the form of apex predators, primarily sharks [32,107] (and references therein). Small MPAs are unable to protect these fish, which swim in and out of the MPA and are thus exposed to fishing for most of the time (since the MPA is a small part of their home range) [230]. Small MPAs can't restore the most damaged part of the fish community, the big fish. Small MPAs can, however, protect spawning aggregations effectively, even though the fish move significant distances going to and from the site [230]. In places where fishing is intense, which is most of the world's coral reefs, not only are the largest fish species mostly gone, but most of the medium size fish (about 30–100 cm length) are also largely gone. In those situations small MPAs can work well, since most of the medium size fish may have home ranges smaller than the MPA. But we need to take care not to give in to shifting baselines, and remember that restoring medium fish is only the first step to restoring coral reefs, and restoring the largest part of the reef biomass in the largest species, needs to be a goal also. Small MPAs are also unlikely to be self-seeding for many fish, and may not be self-sustaining without help from surrounding reefs [165], so good management outside MPAs is also critical. In the Caribbean, the highest fish biomass and large predators are in the largest MPAs [130]. MPAs can benefit fisheries if fishing pressure is heavy, the fish are relatively sedentary, the MPA is large enough, and it is well enough enforced. However, many MPAs are not well designed or effective, just as stock assessments and fisheries regulations can work if implemented properly but have often not worked because they have not been properly implemented. Many MPAs fail because socioeconomic factors are not incorporated into planning and management. Using socioeconomic factors can increase compliance and thus the chance of success in MPAs [353].

Fisheries enhancement is not the only possible benefit of an MPA for villagers, and may not even be the best. Where dive tourism is available, tourism can easily provide far more economic benefit than fishing. Tourism is the world's largest industry, and reef tourism is very large in some areas [354]. It is one of the largest sources of incomes for small islands in the Caribbean. Cozumel, Mexico, which has an economy primarily based on dive tourism in the MPA there, is one of the largest foreign exchange earning areas for Mexico [355] and has been for about 50 years. In the Great Barrier Reef Marine Park, reef tourism was estimated to be about AU$589 million per year while commercial and recreational fisheries were about AU$211-331 million per year [356]. Dive tourism is a large part of the economy of Palau, with shark diving making up 8% of the gross domestic product, and providing 14% of all business tax revenues. A single shark is calculated to be worth US$1.8 million dollars during its

lifetime to the dive tourism industry in Palau, compared to just US$10,000 as the total fisheries value of all sharks in shark dive tourism there [357]. Sharks, humphead wrasse and bumphead parrots are now protected throughout the Palau EEZ (Exclusive Economic Zone). Coral reefs have substantial value for recreation [358], and MPAs enhance that by increasing the large fish that attract divers. Tourists rate high coral cover, structural complexity, fish abundance, and large fish as factors that attract them [186]. The economics of tourism and fisheries may make such systems more viable when the MPA is small or fisheries are small [331].

Dive tourism is non-extractive, and when properly conducted, can have a relatively small ecological footprint [353]. Of course, when improperly conducted, it can have very damaging effects on coral reefs. There are many studies documenting the damage divers and snorkelers can do [353] (and references therein), and resort construction and resort solid and liquid wastes are a problem, plus tourists can increase demand for seafood and thus fishing [75,353]. Dive operators have an incentive to minimize the impact of their divers. Reef tourism isn't available at all locations, and in low-income countries with large amounts of coral reef like Indonesia and the Philippines, there isn't nearly enough dive tourism to provide benefits to fishers in most areas. Also, dive tourism need not benefit local fishers or the traditional owners of the reefs, the local villagers [307]. Dive operators might hire only outside people, and little or no benefits may flow to local fishers. People may feel like the poor and powerless have their source of livelihood confiscated without compensation by the rich and powerful to use exclusively for their recreation, causing resentment. Changes in resource allocation are common side effects of fisheries management tools [359], and equity needs to be a consideration. But dive operators can also choose to hire local fishers and if fishers and villagers receive significant benefits, they may feel they are treated fairly and support the MPA, increasing compliance. For example, several conservation organizations working in Fiji negotiate agreements with fishing villages that if the village does not fish in an MPA area, they will hire a certain number of village people to work in the dive industry and provide several other benefits such as scholarships, hiring enforcement agents, payments of dive taxes into a village bank account, *etc.* [360,361]. MPAs are attractive to divers since they can restore fish populations including the big fish, like the sharks and other large fish that attract divers to Palau. Properly designed and implemented, MPAs can benefit reef conservation, fish stocks, fishers, villagers, and dive tourism simultaneously, but implementation is not easy.

4.6. Dataless Management and Traditional Ecological Knowledge (TEK)

"Dataless management" is the phrase Johannes [235] uses to describe using qualitative information about reef fish to guide management. In many traditional societies, village leaders have managed coral reefs sustainably for hundreds of years without any scientific data. However, fishers had built up knowledge (TEK) of the reefs and the organisms in them that can be superior to the information from conventional resource surveys [362] or knowledge that fisheries biologists don't have [363]. Most but not all TEK is supported by science when tested [363]. Including TEK in management, increases local support [364]. The leaders use that knowledge to decide on management steps to take, like season and spatial closures, taboo species, species that can only be eaten by some people, and so on. One example is traditional knowledge of spawning aggregation times and sites. With that knowledge, village leaders can announce closures, and protect the resource as well as if there was sophisticated data on how many

fish were where and when. Another example is that in the Solomon Islands, some areas are closed when reef fish show an increased flight distance, which makes spear fishing more difficult, but also may indicate heavier fishing pressure [365]. The qualitative information could come from people with traditional knowledge or scientists, and the regulations could be village based or governmental. Key informant interviews would be an important step for managers. "The information that is necessary to evaluate potential replenishment of marine resources relies not on statistical analyses of surveys of the population densities of stock or assessments of the catch-per-unit effort, but on the straightforward observation of whether the big ones are still there" [323]. Scientists might have to be more tolerant of anecdotal evidence, since that is what TEK is [366]. Another example would be the informal knowledge from fishers and observers that documented the fishery for bumphead parrots in Fiji that drove the species to low levels or local extinction on some islands [74]. Information from scientists that bumphead parrots schools were once present in Guam but spear fishing removed them and now they are very rare is another example. That could lead to protection, and knowledge of their movements might lead to protection throughout Guam, but has not so far. Lack of political will, and ardent opposition from fishers, are apparently the key stumbling blocks. But not protecting these fish does not benefit the fishers. If they are nearly extinct the fishers can't catch any, but if they are restored there could be a sustainable catch if catch could be controlled. The knowledge is now sufficient to indicate that most of the largest species of reef fish need to be protected everywhere near people [32], and not just in small MPAs. Once restored, sustainable fisheries for them could be possible. A co-management system, such as a community-based management system with scientist and government assistance, may be able to use "dataless management" to achieve surprisingly good management. Community-based MPAs can allow communities to regain control of the resources they depend on [367]. This could be supplemented with "adaptive management" where management actions could be experimental, and used to find the best management methods for a particular area [207].

4.7. Customary Marine Tenure (CMT)

CMT or Customary Sea Tenure (CST) is the traditional village tenure system of some societies, which was successful at managing fisheries for hundreds or even thousands of years. CMT depends on TEK. CMT is a communal ownership pattern that is inherited [363] but can't be transferred [368] except to offspring or kin [276], a system of rules about who can take what, when and how, plus taboos. Spatial closures are almost always temporary, unlike MPAs, and gear bans have included gill nets, spear guns, cyanide, and root poisons. Catch restrictions are to only take what you need, and not waste any [369]. It is primarily a system of asserting exclusive rights to utilize resources by groups such as villages, clans or families, and while most who study it view its purpose as not conservation [276,370], it often results in conservation, though there are many cases where indigenous peoples have over-exploited resources [369]. Tenure is more like jurisdiction than ownership, it is power to control, but not to buy, sell or trade [368]; it is a communal form of rights-based management [276]. CMT varies greatly from one place to another, and it is unwritten, flexible [368], and constantly evolves [368,369] probably because of the small size of the group making decisions [371]. It can be stronger in some locations than others, with some groups being able to exclude others, but other groups not able to [363]. Places with strong CMT had more conflicts, fewer immigrants from other areas,

were farther from markets, and more dependent on fishing [371]. It is a cost-effective management system [363] because it involves no extra cost; enforcement is local like in community-based MPAs in the Philippines [371]. Compliance is often high, and has been shown in some cases to be higher than in MPAs or community-based management. It has been recommended as an alternative to MPAs, since MPAs have a low success rate [369]. CMT and community-based management both require enabling legislation from central governments [372]. If the state does not recognize CMT there can be conflicts since some people will challenge the right to exclude others [371]. Periodic opening of closed areas (often for feasts) may not be as effective as permanently closed MPAs in conservation [365] or increasing fish catches, but in one study controlled periodic harvests produced increases in abundance and biomass of fished species over control areas [373]. The Vanuatu government recommends that trochus be closed until stocks reach target levels before harvest [373] which is similar to conventional fisheries closures. Fish sizes and biomass were reported in another study to be greater in a temporarily closed area, but it seems unlikely that could be due to the closure since it was only for 6 months [365]. If periodic harvest is uncontrolled and too intense, periodically harvested areas may not show benefits [374]. Modeling studies indicate that rotational openings or periodic harvesting may be able to provide benefits in some circumstances [339,375,376]. However, for sea cucumbers, one proposal is to have the fishery closed most of the time, and then opened briefly for pulse fishing. Stocks would be surveyed before and after pulse fishing to guide management [293]. If they were protected well enough during closure and stocks were low enough to risk Allee effects, the few remaining individuals of these sedentary species could be gathered together to increase fertilization success, to jump start recovery, and the same could be done with *Diadema* in the Caribbean. The complexities of CMT can make development projects difficult [369]. CMT needs to be maintained where it still exists, since there are so few effective alternatives. If it is weak, then co-management [377] or community-based management should be used. Local control often works much better than central control [377]. Hybrid systems such as MPAs plus CMT or community-based plus CMT or species and gear restrictions plus CMT are increasingly common [369]. Setting up MPAs within CMT systems strengthens the MPAs, while setting up MPAs in conflict with CMT system can doom them to failure [276]; MPAs need to work with local culture not against it [353]. Unfortunately, in many places outside pressures such as Western open-access law and market pressures [369] have reduced the ability of this system to sustainably manage fisheries, and in some of the countries with the most reefs, traces of the system are hard to find in some areas [371,367]. High market value can cause serial depletions in CMT areas. Western components of CMT can reduce flexibility, and if CMT is not followed there can be conflicts [369]. Where CMT exists, traditionally managed reserves may do better than co-managed reserves or national park MPAs [378]. Reconstructing successful management practices of the past may help us to build a better future [68]. After a period of decline, community-based and CMT management systems have expanded in recent years [372].

5. Conclusions

Coral reefs face formidable challenges from climate change, and from the difficulties in managing highly diverse reef fisheries. Fishing is one of the greatest of many proximal causes of reef decline, while the ultimate causes appear to be rapid population growth [22,131,245] and economic

development [245]. Solving these problems will not be easy. We must solve these challenges, because the survival of coral reefs as we know them depends on it, and hundreds of millions of people depend on coral reefs for food, shoreline protection, income, and other ecosystem services. We must do a much better job of managing reef fisheries if we are to restore overfished and nearly extinct fish species, maintain and rebuild ecosystem services, and build reef health and resilience to meet the looming challenges of climate change on coral reefs.

Acknowledgments

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