

More, Bigger, Better and Faster: The Future of Marine Protected Areas

CALLUM ROBERTS

Environment Department, University of York, York, YO10 5DD, United Kingdom.

FISHING DOWN THE FOOD WEB AND THE TRANSFORMATION OF CORAL REEFS

Daniel Pauly coined the expression ‘fishing down the food web’ in a highly influential paper published in 1998 (Pauly et al. 1998). In it, he and his colleagues described how the average trophic level of fish recorded in FAO landings statistics was in decline in many parts of the world. They argued that fishing down food webs occurred because predatory species higher in food webs tend to be depleted first, causing fishers to switch to other, more abundant species, which tend to come from lower in the food web. Predators are targeted first because they are often large, bold and voracious, and therefore easy to catch, have firm tasty flesh and are valuable. Over time, overfishing necessitates a succession of such switches as ecological communities are stripped of their predators.

Pauly’s idea has been challenged on several grounds. Essington et al. (2006) contend that fishing takes place through food webs rather than down, so fisheries for species lower in food webs are added while those for top predators continue, albeit at lower levels than before. Sethi et al. (2010) suggest that it is more accurate to view fishing down in economic terms with the most valuable species targeted first and then lower value species added as these are depleted. Since many of the most valuable species are top predators the effect also appears as fishing down the foodweb. Finally, some authors contend that FAO landings statistics show a fishing down phenomenon that is less apparent in biomass surveys of fish at sea (Branch et al. 2010). However, abundant historical data indicate that fishing down the food web cannot be explained away so easily. There is compelling evidence from just about every sea and gulf in the world that large-bodied species, usually predators, have fallen to levels far below historical maxima (Jackson et al. 2001, Roberts 2007). In some places, such as Scotland’s Firth of Clyde, the same effects have been demonstrated using both landings data and biomass surveys (Thurstan and Roberts 2010, Heath and Speirs 2011).

Pauly et al. (1998) expressed fishing down the food web as taking place over time. The same phenomenon can also be observed over space (Figure 1). Where there are geographic gradients in fishing intensity, similar habitats support very different fish assemblages. Within regions like the Caribbean, the differences owe far more to the intensity of fishing than to biogeography or variation in habitat. A few years ago, my wife Julie Hawkins and I counted fish at six islands across the Caribbean. We found that the biomass of predatory fish fell ten-fold from the lightly fished reefs of Bonaire to the intensively exploited reefs of north Jamaica (Hawkins and Roberts 2004, Figure 2). So did the biomass of herbivorous fish, although the decline through sites with intermediate fishing intensities was less steep. Non-target bycatch species like butterfly and angelfish also declined as fishing pressure rose (Hawkins et al. 2007). Within fish families, species declined across the gradient of fishing pressure in sequence of body size. Many had disappeared altogether by the time fishing pressure reached Jamaican intensities, while others were represented only by juveniles that had probably recruited from distant, less-fished reefs (Hawkins and Roberts 2003). Newman et al. (2006) replicated these findings for a different set of study sites in the Caribbean, while Stevenson et al. (2006) showed a similar phenomenon on coral reefs of the Pacific Line Islands. In the Caribbean, Hawkins and Roberts (2004) also saw a gradient of declining coral cover, reduced habitat structural complexity and increasing algal cover as fishing intensities went up. With herbivorous fish ten times less abundant at the most fished sites, this is hardly surprising. But it shows how fishing has effects that cascade through the ecosystem to cause profound differences in structure and function (Estes et al. 2011).

Although it was the least fished site in our study, Bonaire was far from unaffected by fishing. A brief glance at the historical record revealed that Bonaire was once home to abundant reef sharks and several large species of grouper, including the goliath grouper (*Epinephelus itajara*), that were absent by the time of our survey (Hass 1952). They had already fallen victim to overfishing. The baseline from which we inferred the impact of exploitation was not really a baseline at all. The same is true of virtually anywhere else you care to look in the region. Old photographs of fish catches show how big animals like groupers were once far more abundant on Caribbean reefs (McClenachan 2009).

In a fascinating piece of historical sleuthing, Loren McClenachan and Andrew Cooper used the extinct Caribbean monk seal to infer how many fish the region’s reefs might have supported hundreds of years ago (McClenachan and Cooper 2008). They used a wide range of historical sources, including accounts written by pirates, complemented with population modeling to piece together how many monk seals there were in the 17th century Caribbean. Historical observations showed the seal was widespread from the Gulf of Mexico to South America. The authors estimated there were 233,000 to 338,000 seals distributed throughout this range. Such a large population would have required a lot of fish. Based on food intake rates by the ecologically similar Hawaiian monk seal, McClenachan and Cooper calculated that Caribbean reefs would need to have sustained abundances of fish equivalent to 700 to 1,000 g/m² to support all these seals. This is higher than the biomass reported for any Caribbean reef today and similar only to levels attained in remote, uninhabited Pacific reefs like Palmyra (McClenachan and Cooper 2008).

USING MARINE PROTECTED AREAS TO PROMOTE RECOVERY

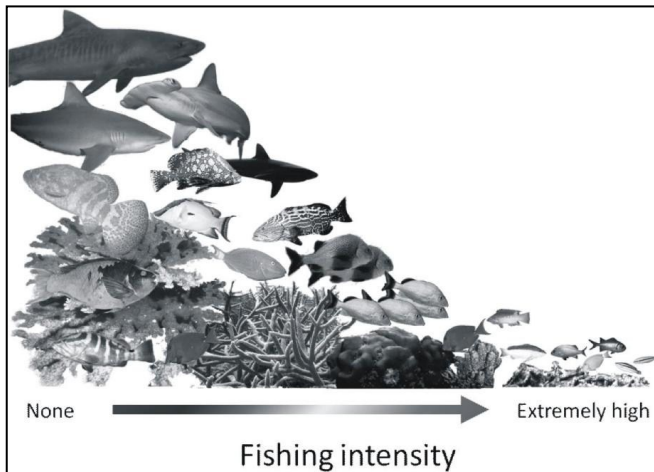


Figure 1. Fishing down the foodweb as seen in spatial comparison of places subject to different fishing intensities.

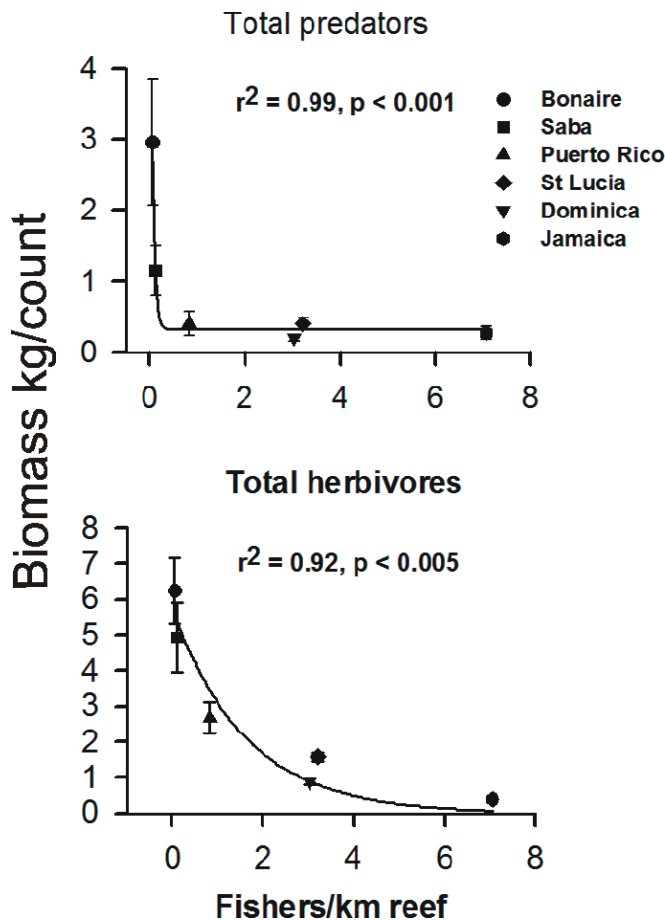


Figure 2. Comparison of total biomass of predatory fish (groupers, snappers and grunts) and herbivorous fish (parrotfish and surgeonfish) across a gradient of fishing intensity. Fish were censused using 15 minute stationary point counts 10 m diameter. See Hawkins and Roberts (2004) for further details.

In the last couple of decades, there has been a revolution in our understanding of fishing impacts on marine life. Some of this stems from research of the kind I have described, in which we infer the effects of fishing from spatial or temporal patterns of decline. But the other line of evidence comes from what happens when we fish less. Protection from fishing, whether in marine protected areas (MPAs), fishery closures, or war-afflicted seas has time and again led to rapid and prolonged increases in biomass, abundance, diversity, and body size of exploited species (see summaries in Gell and Roberts 2003, Lester et al. 2009). Over timescales of decades, the cascading effects of such recoveries feed through to changes in habitats and ecological communities (Edgar et al. 2009). On similar multidecadal timescales there can be recovery of some long-lived, large-bodied species (Roberts et al. 2001; Babcock et al. 2010).

Much of what we know about the performance of MPAs, and much of the rhetoric about them, is based on research in highly protected sites that are closed to most or all forms of fishing. Scientists naturally gravitate towards places where they can expect to get significant results quickly. So there has been a tendency to choose well managed no-take zones as study sites. But such places represent just one end of the spectrum of protection afforded by MPAs. Less-protected sites tend to be the norm. When MPAs are proposed, they often generate intense controversy because they can affect what people do and where they do it. The arguments get especially heated over fishing as livelihoods are involved. Having made public commitments to establish MPAs (e.g. at the 2002 World Summit on Sustainable Development) politicians are loath to abandon them altogether when the going gets rocky. Instead, they usually compromise on the level of protection given, with the result that many MPAs actually offer very little real protection.

How good are partially-protected MPAs in reversing the decline of ocean wildlife? Here Pauly's fishing down the food web idea is useful to frame the answer. While fishing drives ecosystems on the downward trajectory shown in Figure 3, MPAs can be seen as a countervailing force to push them back. How far back you can travel along the gradient of overfishing depends on how much protection is given. Certain fishing gear restrictions, such as a ban on spearfishing, or use of a larger mesh on nets, might take an intensively fished habitat back only a little way, producing modest increases in biomass and size of some fish, and possibly some improvement in the cover of biogenic habitat. Greater protection within MPAs, perhaps exclusion of bottom trawling or a ban on use of fish pots, can produce greater recovery. The variety and size of fish will continue to grow and species that were rare or absent may begin to reappear, while habitats may recover to a greater degree.

Full protection from all exploitation will push ecosystems back to a more intact state, but in most places this will fall short of full recovery to a pristine state. For one thing,

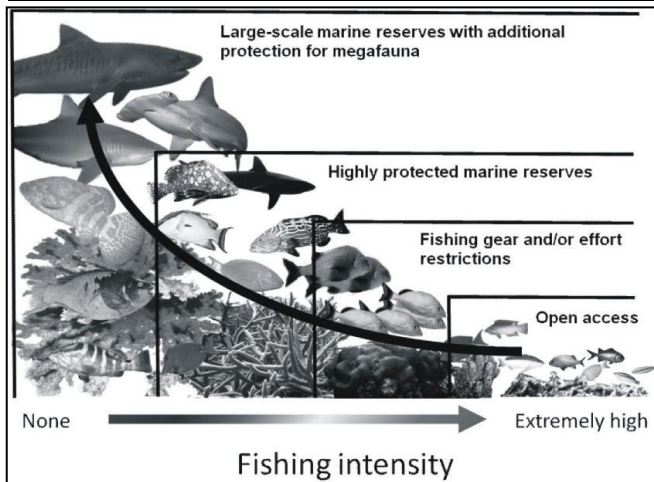


Figure 3. Protective measures seen as a countervailing force to fishing down the foodweb. Increasing protection moves ecosystems to the left on the recovery trajectory shown by the black arrow.

there are many human stresses that MPAs cannot fully exclude, such as pollution and climate change. But there is also the problem that most MPAs are small – in the range of a few hectares to a few square kilometres. Although on close scrutiny, many species that appear to be highly mobile turn out to be less so (Neat et al. 2005, Norse et al. 2005, Roberts and Mason 2008), nonetheless, large animals like sharks can be expected to gain only partial protection from small MPAs. Recovery might progress farther back toward the pristine end of the spectrum in very large and highly protected MPAs, but the evidence for this remains sparse mainly due to the paucity of such places and the rather limited time they have been in existence measured in terms of generations of large-bodied animals. In addition, such places rarely exclude people. Research from the Great Barrier Reef found that reef sharks were much more abundant in strict protection zones from which people were excluded than in the no-take ‘green’ zones that allow access but prohibit fishing (Robbins et al. 2006).

Finally, even highly protected marine reserves cannot resurrect the dead. Some species in the Caribbean have been extirpated from large parts of their former range. One of the only large groupers I saw in years of fish counting on the intensively exploited reefs of St. Lucia, was a yellowfin (*Mycteroperca venenosa*) that was stuck in a lost fish trap (Figure 4). Ironically, its life had been spared by being caught and it had grown fat on a stream of unwary fish that found their way into the same trap. To bring back species like goliath and Nassau groupers, giant clams, sawfish or staghorn coral, the only options are to wait and hope for some lucky recolonization event or events to kick start recovery or to reintroduce species that have been lost.

Moving exploited ecosystems back along the recovery trajectory by giving them protection is more than simply a matter of aesthetics. It makes economic sense too. In Figure

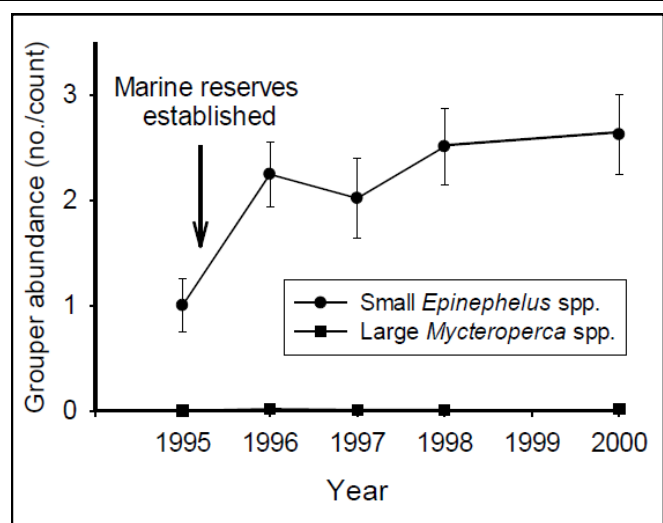


Figure 4. Change in abundance of large and small groupers in marine reserves on reefs of St. Lucia after their establishment in 1995. Fish were censused using 15 minute stationary point counts of 10 m diameter.

5, I show some general relationships between economic values of coral reefs and their ecological state based on my reading of the literature and personal experience of such reefs. Figure 5a shows a curve of fisheries catch value superimposed on the fishing down the food web diagram. Total catches peak around the middle of the gradient of fishing intensity because fish communities at this point still sustain reasonably high biomass and this is made up of mid-size species that have relatively high turnover rates. However, the peak monetary value lies to the left of peak catch because catches there are dominated by large-bodied, higher value animals. To the far right, degraded reefs sustain a fraction of the biomass of fish and it is made up mostly of small-bodied, less desirable, low value species. Fishery catch values represent only one element of the value of natural ecosystems. Figure 5b shows a curve for the tourism value of reefs with the fishery catch value curve rescaled relative to it. Tourism values far outstrip fishery values for all ecosystem states. However, tourism value falls off as reefs become degraded, although they never fall to zero because there are many tourists for whom reefs are nothing more than a warm bath. Even degraded reefs can fulfill that role perfectly well! Figure 5c shows tourism and fishery values rescaled to a third line, this time the value of the sum of other ecological services that reefs perform, such as protection from coastal erosion and sea level rise, water filtration and purification, and carbon sequestration. When reefs are healthy and sustain prolific fish and coral communities, these values likely outstrip those from tourism and fisheries, but they fall off rapidly as coral reefs degrade toward the right hand side of the fishing down trajectory.

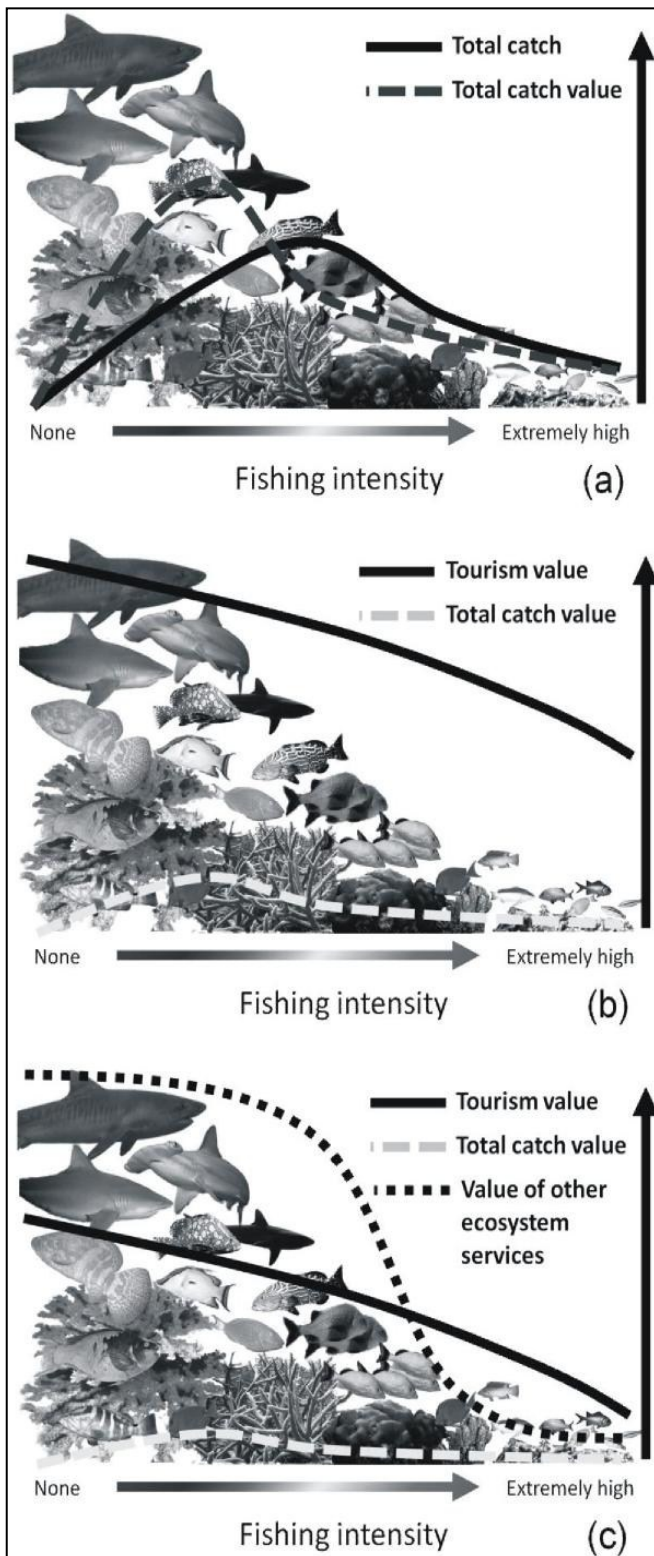


Figure 5. Relationships between coral reef ecosystem values in relation to ecosystem state.

MORE, BETTER, BIGGER AND FASTER MPAS

So far I have concentrated on the effects of fishing on marine life and habitats. But they are not the only effects we have. Alongside direct impacts such as overfishing, pollution and introduced species, climate change and ocean acidification now form a backdrop of growing environmental stress. The combined influence of multiple stresses is almost certainly greater than the sum of individual impacts, since effects propagate and magnify through ecosystems due to species linkages. For some habitats, such as coral reefs their entire future is at risk (Veron et al. 2009). MPAs have been widely promoted as a means of safeguarding marine life, or at least alleviating local impacts while humanity struggles to reduce greenhouse gas emissions. Indeed, they are central in efforts to protect biodiversity under the Convention on Biological Diversity (CBD). In 2010, the CBD agreed to expand global coverage of MPAs to 10% by 2020, a six-fold increase on the present 1.7% (as of 2011).

As I have shown in this paper, the ability of MPAs to fulfill this biodiversity safeguarding role is compromised by insufficient protection and small size. When it comes to MPAs, you get what you pay for. Weak protection can only produce small benefits. Strong protection produces strong effects and will promote much greater recovery and therefore benefits. Weakly protected MPAs will do little to increase the resilience of marine ecosystems to human stresses and global change. Strongly protected MPAs can offer a lot. They can increase population sizes much more, which means greater biomass, more offspring, greater connectivity of populations across seascapes, less likelihood of extinction and greater rates for key ecosystem processes such as grazing, water filtration, and growth of biogenic habitats.

In my view, highly protected MPAs are critical to sustaining marine ecosystems in diverse, productive and valuable states while the world struggles to feed billions of new people and bring carbon dioxide emissions under control. To achieve this, and to meet the demanding CBD goal of 10% by 2020, we will have to get much better at establishing this kind of MPA. Scotland offers a perfect example of the hurdle we must overcome. Arran is a small island in the Firth of Clyde that is home to 5000 people. In the 1990s a small group of islanders noticed that marine life was being destroyed by overfishing and use of destructive gear like scallop dredges. They began a campaign to establish a small no-take zone – just 2.4 km² – in one of the island's bays. For years their pleas fell on deaf ears in government and provoked hostility from commercial fishers, but they wouldn't give up. In 2008 after more than 15 years of unceasing effort they got their no-take zone. 2.4 km² is a tiny drop in the ocean. The oceans will not recover if all such victories must be so hard won. We have to get better at establishing bigger and more highly protected MPAs much faster if we are to get anywhere close to the CBD goal. And 10% coverage of MPAs is only a milestone on the way to effective MPA

networks for coasts and oceans. The best available science indicates that we will need to step up coverage of MPAs to 30% or so if they are to make a meaningful contribution to resilience and productivity at regional and global scales (Gell and Roberts 2003).

In saying this, I don't mean to denigrate efforts like those of the people of Arran. In fact just the opposite. Their no-take zone and the small MPAs of hundreds of other communities like them scattered across the world offer hope and inspiration. They provide working examples of protection that can motivate others to do the same. I see them as a potential springboard to the establishment of more extensive and more highly protected networks in future. Having won the arguments over the legitimacy and necessity of protection, they reshape the political climate so that future MPAs should not face the same obstacles. Probably every country will need a few cases hard fought by similarly stubborn communities as the people of Arran. Every place is unique culturally, as well as different biologically, and I have noticed that people are reluctant to accept any experience other than their own. The slow progress can be disheartening to those engaged in such struggles, but they can take heart that as pioneers they are blazing a trail so that others can travel more easily.

ACKNOWLEDGEMENTS

I would like to thank the organising committee of the 63rd GCFI Conference for their kind invitation to give this keynote presentation. I would also like to thank Julie Hawkins for her help, encouragement and keen insights offered throughout the years.

LITERATURE CITED

- Babcock, R.C. et al. 2010. Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. *Proceedings of the National Academy of Sciences* doi/10.1073/pnas.0908012107
- Branch, T.A. et al. 2010. The trophic fingerprint of marine fisheries. *Nature* **468**:431-435.
- Edgar, G.J., N.S. Barrett, and R.D. Stuart-Smith. 2009. Exploited reefs protected from fishing transform over decades into conservation features otherwise absent from seascapes. *Ecological Applications* **19**:1967-74.
- Essington, T.E. et al. 2006. Fishing through marine foodwebs. *Proceedings of the National Academy of Sciences* **103**:3171-75.
- Estes, J.A. et al. 2011. Trophic downgrading of Planet Earth. *Science* **333**: 301-306.
- Gell, F.R. and C.M. Roberts. 2003. Benefits beyond boundaries: the fishery effects of marine reserves and fishery closures. *Trends in Ecology and Evolution* **18**:448-455.
- Hass, H. 1952. *Diving to Adventure*. Jarrolds Ltd, London, United Kingdom.
- Hawkins, J.P. and C.M. Roberts. 2003. Effects of fishing on sex-changing Caribbean parrotfishes. *Biological Conservation* **115**:213-226.
- Hawkins, J.P. and C.M. Roberts. 2004. Effects of artisanal fishing on Caribbean coral reefs. *Conservation Biology* **18**:215-226.
- Hawkins, J.P., C.M. Roberts, F.R. Gell, and C. Dytham. 2007. Effects of trap fishing on reef fish communities. *Aquatic Conservation: Marine and Freshwater Ecosystems* **17**:111-132.
- Heath, M.R. and D.C. Speirs. 2011. Changes in species diversity and size composition in the Firth of Clyde demersal fish community (1927 - 2009). *Proceedings of the Royal Society of Britain*. published online 6 July 2011. doi: 10.1098/rspb.2011.1015.
- Jackson, J.B.C. et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* **293**:629-638.
- Lester, S.E. et al. 2009. Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series* **384**:33-46.
- McClenachan, L. 2009. Documenting loss of large trophy fish from the Florida Keys with historical photographs. *Conservation Biology* **23**:636-43.
- McClenachan, L. and A.B. Cooper. 2008. Extinction rate, historical population structure and ecological role of the Caribbean monk seal. *Proceedings of the Royal Society of Britain* **275**:1351-1358.
- Neat, F.C. et al. 2005. Residency and depth movements of a coastal group of Atlantic cod (*Gadus morhua* L.). *Marine Biology* DOI 10.1007/s00227-005-0110-6.
- Norse, E.A. et al. 2005. Place-based ecosystem management in the open ocean. Pages 302-327 in: E. Norse & L. Crowder (eds.) *Marine Conservation Biology: The Science of Maintaining the Sea's Biodiversity*. Island Press, Washington DC, USA.
- Newman, M.J.H., G.A. Paredes, E. Sala, and J.B.C. Jackson. 2006. Structure of Caribbean coral reef communities across a large gradient of fish biomass. *Ecology Letters* **9**:1216-1227.
- Pauly, D., V. Christensen, J. Dalsgaard, R. Froese, and R. Torres. 1998. Fishing down marine food webs. *Science* **279**:860-863.
- Robbins, W.D., M. Hisano, S.R. Connolly, and J. H. Choat. 2006. Ongoing collapse of coral-reef shark populations. *Current Biology* **16**:2314-2319.
- Roberts, C.M. 2007. *The Unnatural History of the Sea*. Island Press, Washington, DC USA.
- Roberts, C.M, J.A. Bohnsack, F.R. Gell, J.P. Hawkins and R. Goodridge. 2001. Effects of marine reserves on adjacent fisheries. *Science* **294**:1920-1923.
- Roberts, C.M. and L.C. Mason. 2008. Return to Abundance: A Case for Marine Reserves in the North Sea. WWF-UK. <http://www.wwf.org.uk/filelibrary/pdf/marinereservesnorthsea.pdf>.
- Sethi, S.A. et al. 2010. Global fishery development patterns are driven by profit but not trophic level. *Proceedings of the National Academy of Sciences* **107**:12163-67.
- Thurstan, R.H., S. Brockington, and C.M. Roberts. 2010. The effects of 118 years of industrial fishing on UK bottom trawl fisheries. *Nature Communications* **1**:15 | DOI: 10.1038/ncomms1013.
- Thurstan, R.H. and C.M. Roberts. 2010. Ecological meltdown in the Firth of Clyde, Scotland: two centuries of change in a coastal marine ecosystem. *PLoS ONE* **5**(7): e11767. doi:10.1371/journal.pone.0011767.
- Veron, J.E.N. et al. 2009. The coral reef crisis: The critical importance of < 350 ppm CO₂. *Marine Pollution Bulletin* **58**:1428-1436.