

FISHERIES MANAGED TO REBUILD ECOSYSTEMS? RECONSTRUCTING THE PAST TO SALVAGE THE FUTURE

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Abstract. This paper presents the case for adopting ecosystem rebuilding as the goal of fisheries management. Movement toward this goal may represent the only hope for fisheries, as we know them, to exist 50 years in the future alongside essential services provided by marine ecosystems. First, I review archaeological, historical, and recent evidence that bears witness to a long, dismal record of overexploitation. Second, I examine the ecological effects of overfishing on aquatic ecosystems. Fish with life histories and spatial behavior inimical to harvesting are selectively removed, both within and among species. The loss of keystone species and the replacement of high-value, demersal resources with pelagic, rapid-turnover, low-value species shifts the nature of ecosystems, evidenced by accelerating local extinctions and a worldwide decline in trophic level. Disconcertingly, harvest limits that appear safe by single species evaluation can engender ecosystem changes that are hard to reverse. Driven by a progression of clever human harvest technologies, three ratchet-like processes have brought about episodes of depletion. “Odum’s ratchet” is ecological in nature, comprising depletion and local extinction. “Ludwig’s ratchet,” economic in nature, is a positive feedback loop between increased catching power and serial depletion, driven by the need to repay borrowed money. “Pauly’s ratchet” is cognitive, shifting the baseline of what each generation regards as primal abundance and diversity. Third, a rebuilding policy goal is distinguished from that of sustaining current catches and biomass, since the baseline can refer to present misery. In this sense, present policies can inadvertently foreclose future options for the generation of food, wealth, and services from ocean resources. A policy to rebuild ecosystems can reverse this trend and maximize economic value in tomorrow’s markets, where supply will vastly outstrip demand for high-quality fish products. Fourth, I outline a novel methodology, termed “Back to the Future,” that can implement a goal of ecosystem rebuilding. Models of past ecosystems are reconstructed using information about the presence and abundance of species from historical documents, archaeology, and local and traditional environmental knowledge (LEK and TEK). Economic evaluation compares past with present and alternative ecosystems. “Back to the Future” gives the TEK of aboriginal and indigenous peoples a valuable, direct function in resource management. Finally, I discuss two practical management measures, paralleling recent developments in terrestrial reconstruction ecology, the implementation of large no-take marine reserves, and the reintroduction of high-value species that were formerly endemic.

Key words: biodiversity; ecosystem rebuilding; fisheries management and sustainability; fisheries policy; harvest refugia; history of fisheries; mass-balance models; restoration ecology.

*Time present and time past
Are both perhaps present in time future
And time future contained in time past*

—T. S. Eliot, *Burnt Norton*, 1935

INTRODUCTION

The existing paradigm in fisheries science is the sustainability of single species. This science is not wrong, but, just as Newtonian physics is not as useful as relativity or quantum theory in evaluating the cosmological and subatomic nature of our universe (Hawking 1988), it fails to answer the most pressing questions of

our time. These questions concern the changes wrought by fishing upon the composition and nature of aquatic ecosystems, and how fishing may continue without causing irreversible change.

Something is clearly amiss with the management of fishery resources. It is a sad paradox that a branch of applied ecology with a 50-yr history of quantitative predictive theory (Sissenwine and Rosenberg 1993, Beverton 1998, Holt 1998) is associated with such a poor track record of numerous and repeated fishery collapses that the public anticipates the next disaster, and its economic and social damage (e.g., Greenpeace 1993, Parfit 1995, Safina 1995, M. Harris 1998). Every year, fishery collapses continue to take fishery scientists by surprise (e.g., Chilean horse mackerel; Anonymous 1998). If this were engineering and we were dealing with bridge collapse, materials and methods

Manuscript received 18 January 2000; revised 3 April 2000; accepted 4 April 2000.

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would have been questioned long ago. Yet many fishery scientists hope that steady improvements to the ecological, mathematical, or social sciences will enable the corner to be turned, and that it will be all right next time.

As evidence mounts of ineffectual fishery management, we need good analytical and predictive tools more than ever if there are to be fisheries as we know them 50 yr in the future. Fishery resources stand challenged by factors acting on a global scale throughout the world's water bodies (Mace 1997): progressive and high-variance climate change; innovative, potent, and overcapitalized catching technology; massive, mounting, and unprecedented excess of demand over supply; and unparalleled social pressure for jobs, recreation, sectoral equity in allocation, and the restoration of indigenous people's rights.

Currently, several candidate approaches offer salvation. First, quantitative ecologists tell us that we should identify, quantify, and make explicit all sources of uncertainty, a trend culminating in Bayesian models (e.g., Punt and Hilborn 1997) and statistical decision analysis (e.g., Peterman et al. 1998). If such modeling is performed thoroughly and properly, and the stock response monitored, it is argued that all will be well with fish resources (e.g., Rosenberg et al. 1993, Sainsbury 1998). Second, economists have signaled that the creation of property rights, or marketable harvest rights, will ensure long-term sustainability, reduce conflict, and fund management, enforcement, and long-term research on the resource free of public subsidy (e.g., Hanneson 1998). If we set up marketable harvest rights thoroughly and properly, all will be well with fish resources (Christy 1997). Third, social scientists tell us that if only we would let the stakeholders and local coastal communities manage the resource themselves (see contributions in Pinkerton 1989), there would be little overfishing from the responsible kin of fisherfolk living in harmony with the resource they exploited for generations before large-scale interests came in from outside and spoilt things. If we allow this traditional social system to thrive, all will be well with fish resources. Each of these three solutions are fine-tuned by their proponents (e.g., uncertainty: Hilborn and Walters 1992, Schnute and Richards 1994; economics: Neher 1996; social sciences: Matthews 1995, C. Harris 1998, Jentoft 1998), and occasionally we see combinations (e.g., Fujita et al. 1998). But the evidence suggests that none of them will solve the problem.

In this paper, I argue that our overall record of failure has been an inevitable consequence of adopting the wrong management goal, not the result of incorrect or inappropriate techniques in themselves (although specific mistakes can easily be identified in many individual cases: e.g., the Newfoundland cod collapse; Walters and Maguire 1996). In particular, I contend that the goal of sustainable yield of single species in a fishery is a fundamental mistake, and present evidence that

the ecological effects of fishing on aquatic ecosystems precludes the long-term survival of most present-day fisheries, even when all the criteria of conventional sustainability goals are met. Previous authors have argued persuasively that the goal of sustainable yield from aquatic resources has never been achieved (Ludwig et al. 1993). Under existing goals, I suggest it is neither achievable nor desirable, were it possible.

The paper commences with a review of the historical impacts of fishing on aquatic ecosystems, describes the nature and reversibility of these ecological effects, examines three ratchet-like processes that make restoration difficult, and then presents a policy goal that could allow valuable fisheries to coexist with healthy, intact natural resource ecosystems (in the sense of Spurr 1969, and Mangel 1991). I conclude with a review of a practical assessment and planning framework, termed "Back to the Future," which could enable this goal to be achieved.

THE HISTORY AND IMPACTS OF HUMAN FISHING

Terrestrial experience suggests that we should not be particularly surprised by the effects of hunting. For terrestrial mammalian megafauna, such as the mammoth and associated specialized carnivores and scavengers, a pulse of late Pleistocene extinctions throughout the world coincides with the advent of cooperative hunting behavior and technology as modern humans spread around the globe (Martin 1967, 1984, Stringer and McKie 1997, Ward 1997). In the past 100 000 yr North America has lost 73%, South America 79%, and Australia 86% (Flannery 1990) of endemic genera of terrestrial megafauna. Few megafauna survived; some proboscids, red kangaroos, and others with life histories that could withstand human hunting or that lived in remote habits hostile to humans. The horse, aurochs, and both new and old world camels survived being hunted by humans only after domestication, while both horse and camel appear to have been exterminated in North America (Martin 1984). Evidently, cooperative hunting by humans decimated large mammal biodiversity, which in turn affected dependent species and habitat structure. But a recent symposium volume reviewing the determinants of extinction rates (Lawton and May 1995) treats the extinction of the Pleistocene megafauna as a side issue (Ehrlich 1995:220–221), as though the eminent bird, insect, and plant ecologists at the meeting had forgotten about keystone species. In fact, with the exception of asteroid impact on Earth, anthropogenic influences overwhelm the natural ecological processes of extinction, and the same volume reports them as four orders of magnitude greater than that seen in the fossil record (May et al. 1995). Later in human history, early attempts at agriculture helped to create many of the world's deserts. Our species has "terraformed" major land areas of the planet through species loss and habitat change.

Turning to the oceans, although aquatic megafauna

seemingly escaped the late Pleistocene extinctions, the dismal history of whaling almost allowed them to catch up in the past two centuries. Using early hunting technology, coastal whales, such as grey and right whales, were decimated in the last century, while high-technology whaling since the 1960s brought even oceanic baleen whales to the verge of extinction. It does appear that early humans may have wiped out some aquatic keystone species. Soon after the human colonization of North America (Hoffecker et al. 1993), Steller's sea cow, *Hydrodamalis gigas*, described by Steller (1751) as gentle, trusting, and unable to submerge, vanishes from the fossil record along the Pacific coast (Domning 1978). Surviving to the 1750s only in the uninhabited Komandorskiye Aleutian Islands, it was wiped out as fur trader's food within 30 yr. Moreover, prompted by Steller's reports of their amazingly dense fur, in less than one hundred years, sea otters, keystone species in coastal kelp forests (Simenstad et al. 1978), were rendered almost extinct throughout the North Pacific. Giant sea otters existed in the North Pacific in the Pleistocene (Reidman and Estes 1990), and may have also succumbed to human hunting, or been impacted by the demise of the sea cow.

Early hominids gathered shellfish, but the first known capture fishery using tools dates back to 90 000 BP, when humans living on the banks of what is now a tributary of the Congo river carved clever and effective harpoons out of deer antlers (Yellen et al. 1995), and speared giant catfish during their annual migration and spawning. The earliest evidence of cooperative human hunting is wooden spears found implanted in horses in Europe (40 000 BP, Dennell 1995), but spearing fish was likely a communal activity. Nets, for which social cooperation is essential, were first invented some 35 000 yr ago in southern Europe to capture small terrestrial game animals such as hares and foxes (Pringle 1997), but were probably very soon adapted for fishing. Fishing technology such as nets, hooks, and spears spread rapidly with early modern humans around the globe. For example, characteristic double recurved "flatfish" fishing hooks and multipoint fishing spears are found both among Australian Aborigines (Flood 1995) and the coastal peoples of the North American Pacific (Stewart 1977). Like the wheel, these clever devices needed to be invented only once. From those times on, humans have progressively devised new technologies to increase the effectiveness of cooperative hunting for fish.

Although less well-documented and more difficult to study than large terrestrial mammals, archeological evidence is suggestive that technological advances have been associated with dramatic reductions in biodiversity and progressive depletion of fish populations, albeit over long periods of time. Casteel (1976) and Leach and Boocock (1995) have described techniques using fish skeletal parts, preserved in the middens of ancient human settlements, to trace catches and an-

thropogenic impacts on exploited ichthyofauna through changes in diversity, distribution and growth. For example, summarizing archeological evidence and historical birch bark documents from sites in the former Soviet Union, Casteel shows that, not only did harvested sturgeon and salmon decline in size, but sterlet, bream, rudd, large catfish, and in some locations over 40% of the original fish species, disappeared over 5000 yr of fishing. Effects of changing climate and seacoast location could, in general, be distinguished from the effects of fishing.

At one of the first human settlements on Cyprus, middens dated at 8000 BP revealed catches of very large groupers and other fish, from a fauna previously unexploited by humans (Desse and Desse-Berset 1994). Later fish were smaller in size. Fish fauna diversity and size decreased over 12 000 yr of fish remains in an Andalusian cave (Morales et al. 1994). Reductions in mesh size over 100 yr in nets recovered from a late Roman site on the Red Sea (Wendrich and van Neer 1994) were the result of local overfishing, and species caught today in small mesh nets are rare in the archaeological material. In middens on St. Thomas, Virgin Islands, reef fish size decreases while the mean trophic level of the catch falls from 3.78 in 1500 BP to 3.6 in 600 BP (Wing 1994; E. S. Wing, *personal communication*).

The successive impact of fishing technology on aquatic ecosystems exhibits a continuity from ancient to historical times. For example, using only hemp rope and net, the Mediterranean trap for bluefin tuna (*tonnara*) was ingenious technology to catch fast, giant fish, devised as early as 3000 yr BP. Documents from the Graeco-Persian wars mention slaughtering warriors like tuna in a tonnara (Sarà 1990). Catching fish in the tonnara is known in Italian as a *mattanza*, or slaughter (French: *la madrague*). The trap consists of many kilometers of netted walls designed and carefully positioned to deflect tuna migrating along the coast. The nets lead the tuna into a series of gated chambers, where they are herded from one to the next by fishers waving palm leaves. The last chamber is the *camera della morte* (death chamber). When this chamber is full, the whole fishing village comes out, in a ritualized procession of boats, to manually lift the netting floor, and gaff the flailing tuna. Formerly, as many as 300 hundred giant fish were caught at a time. The technology was preserved through the Dark Ages by Arabs, and in Sicily many words associated with the *tonnara* are Arabic (e.g., *rais* for the chief of operations; Sarà 1990). The traps, in different designs for windward and leeward, and for autumn and spawning migrations, were in use throughout the Mediterranean from Spain to Turkey. A large expansion in *tonnara* occurred in the 1800s as a result of the invention of canning. This depleted stocks so that *tonnara* in Italy were reduced from 54 in 1890 to only about 10 by 1919 (Cushing 1987). Today there is only one, operated annually for tourists at Trapani,

Sicily. The reason for the recent reduction is that bluefin tuna are now even further depleted (the reported Mediterranean catch doubled in one year between 1993 and 1994), attracting calls for endangered status (Mooney-Seus and Stone 1997) like the southern bluefin tuna species (Polachek et al. 1995, Traffic Oceana 1996). This has been the result of two recent technologies that can catch even more fish from depleted stocks: first, hydraulic-powered purse seines mounted on freezer vessels equipped with spotter planes and side-scan sonar, and second, high-technology autobaited deep long lines. These expensive fishing technologies have been driven by high prices of sashimi on the Japanese market. The problem has been compounded by management inaction caused by spatial uncertainties in the data and worrisome underreporting. Alarming, a recent high-technology sonar survey in the Mediterranean found numbers of overwintering giant bluefin tuna "slumbering" in cold, deep water off northern Morocco, where they are now being systematically fished (S. des Clers, *personal communication*).

A second example illustrates the now familiar theme. Local herring stocks (*sensu*: Iles and Sinclair 1982), including the now-extinct Scania stock in the Western Baltic upon which a substantial portion of the wealth of the north European Hanseatic League of cities was based (Hardy 1956, Cushing 1987), were likely eliminated by improvements in netting technology. A fascinating historical example of the spread of driftnet technology for catching herring in Scotland in the early 1800s is portrayed faithfully in the novel *The Silver Darlings* by Neil Gunn (1941). By the end of the period, inshore stocks have gone (see Sahrhage and Lundbeck 1992).

A sad, recent parallel with ancient overfishing is the depletion and collapse of the Newfoundland cod, which high-technology trawlers brought to the verge of extinction in just one decade (Walters and Maguire 1996, M. Harris 1998). Five centuries ago this cod stock was so astoundingly abundant that it was a major factor in the colonization of the North America by Europeans (Kurlanski 1997). Now, the marine ecosystem is so altered by the loss of the cod that its recovery is in doubt, even in the absence of fishing (Pauly et al. 2000).

This long and miserable record argues strongly that overfishing is responsible for the observed changes. Are the examples given here carefully selected to support the case? The literature that specifically links archaeological finds to fisheries is not extensive, and I found no historical examples where fishing has a neutral or positive effect on an aquatic ecosystem. It is, however, not hard to find fishery scientists who are willing to argue that fishery collapses are driven mainly by climate or environmental changes (e.g., Sinclair et al. 1997). In a few archaeological cases, environmental effects can be partitioned from the effects of fishing. For example, climate-induced incursions of colder Atlantic water species into the Mediterranean are reflected

in the fish remains found in an Andalusian cave midden (Morales et al. 1994), but this can be separated from the effects of species and size shifts. Equally, in the recent Newfoundland cod fishery collapse, there is now overwhelming evidence that the chronic effects of overfishing, and not sudden climate change, was the culprit (e.g., Hutchings 1996). Climate may well shift, and we can argue about the mix of causes in a specific case, but this does not absolve fisheries of causing major problems.

ECOLOGICAL PROCESSES CAUSED BY FISHING

Human fishing alters the structure of aquatic ecosystems in ways that are only just beginning to be understood (Apollonio 1994). In many large commercial fisheries, over 80% of the mature fish are caught each year: Such heavy fishing alters ecosystems, perhaps irreversibly. Fisheries can meet rigorous sustainability criteria from single species stock assessment, yet may still alter ecosystems unacceptably. Five ecological processes may be identified (see also Pitcher and Pauly 1998).

First, life history characters may be altered. Large, long-lived fishes, with low natural mortality rates are selectively removed from the system (Apollonio 1994, Musick 1999a). This process shifts the composition of the fish fauna within taxa, among the fish species in a genus, and, insidiously, among genotypes within species. When natural predators do the same thing, the process shapes an evolutionary arms race between predators and prey. But we have virtually ignored the evolutionary impact of human fishing (Policansky 1993, Policansky and Magnuson 1998), which is surprising, given that we now understand how natural selection produces remarkably rapid evolutionary responses (birds: Weiner 1994, fish: Houde and Endler 1990). Authors have noted that shifting the community composition of fish species by fishing seems to reverse the course of evolution (Pauly 1979, Parsons 1992, Apollonio 1994). Large, long-lived organisms, with relatively stable reproduction, such as groupers or cod, are replaced by short-lived animals with more volatile population dynamics, such as sardines or scads. Parsons (1996) extends this by suggesting a replacement by simpler, lower energy organisms within each trophic guild. An extreme form of this is seen when predatory jellyfish replace fish predators.

Second, habitat degradation can be caused by fishing gears. Bottom trawls eliminate fragile structures like clearcutting forests (Watling and Norse 1998). Trawls destroy "oyster reefs" and sponge beds (Young and Glaister 1993, Pauly et al. 1996, Lenihan and Peterson 1998) created over long time periods by sessile planktivores and detritivores. In undisturbed systems, demersal fish either feed on these organisms, feed on their predators, or take refuge in the benthic architecture. Hence, degradation of this habitat acts to redirect carbon flows away from benthic toward pelagic systems,

thereby adding to the effects of anthropogenic nutrient inputs.

Third, the reduction of predatory pressure from fewer benthic fish may lead to an increase of small pelagic fish and squids. When these pelagic resources become available for exploitation they may mask the decline of demersals (Pauly and Murphy 1982, Daan and Sissenwine 1991). The South China Sea and Gulf of Thailand provide examples of such a shift. Catches of valuable demersal fish such as croakers, groupers, and snappers (Sciaenidae, Epinephelinae, Lutjanidae) have been replaced by small pelagics used for animal food, and invertebrates such as jellyfish and squids (Beddington and May 1983, Boonyubol and Pramokchutina 1984, Silvestre and Pauly 1997, Pitcher et al. 1998). A shift from groundfish to cephalopods has been documented worldwide (Caddy and Rodhouse 1998).

A fourth biological mechanism is a trophic cascade (Carpenter et al. 1985) through which increased catches of small pelagic fish reduce food for larger piscivores, which then decline further. More small prey are released for capture by a fishery that is forced to target species even lower down the food web. In a similar way, fisheries compete for forage fish with other species, especially marine mammals (e.g., Trites et al. 1997).

A fifth effect of overfishing is to reduce the predictability of exploited aquatic ecosystems. Plankton blooms may be made more likely by removing large numbers of fish from a marine ecosystem, in addition to being generated in a "bottom up sense" by nutrient enrichment. Parsons (1996) describes quasi-chaotic cycles of average annual zooplankton/phytoplankton biomass, where small differences in the timing and characteristics of reproduction act as chaotic attractors, characteristic of different oceans. Hastings and Powell (1991) show that the removal of fish plankton predators can destabilize ecosystems, causing wild oscillations of irregular phytoplankton blooms. Through cascade effects, this could spread chaotic instability to higher levels in the ecosystem.

Contemporary examples of these processes abound. For example, the immense population of walleye pollock (*Theragra chalcogramma*) in the North Pacific may be a result of the exploitation and virtual extinction of baleen whales (National Resource Council 1996). The great expansion of trawl fisheries in the 1980s, aimed at harvesting pollock, may have exacerbated the situation because large pollock would otherwise cannibalize juveniles, which compete with whales for food. Moreover, in the Bering Sea there are reports of a huge jellyfish increase (Brodeur et al. 1999), a worrisome trend also documented from the South China Sea (Saeger 1993) and the Adriatic (Avian and Sandrin 1988), and which has been amplified to disaster level for an introduced jellyfish in the Black Sea (Shushkina and Vinogradov 1991, Zaitsev 1992).

From about 12 million tonnes in 1950, the world

marine fish catch expanded to an official figure of over 80 million landed tonnes (Food and Agriculture Organization 1997). To this, we must add at least 27 million tonnes of discards at sea (Alverson et al. 1994), and illegal and unreported catch. Where snapshots are available, unreported catch seems about equal to the recorded amount, except where enforcement includes onboard observers, or the chances of detection in relation to rewards are high (Beddington et al. 1997). The "true" world fish catch (Pauly 1996) is therefore likely to be in the region of 120–140 million tonnes, perhaps even as high as 160 million. Moreover, about 30% of all primary production in coastal seas is sequestered by fishing (Pauly and Christensen 1995), only slightly lower than the equivalent rate on land (Ehrlich 1995). This indicates little room for the expansion of fisheries.

On this global scale, the historical effect of fishing since 1950 has been dramatically illustrated by plotting the mean trophic level of fish catch against time (Pauly et al. 1998a). The catch of each species is weighted by its trophic level, obtained from ecosystem modeling (available in "Fishbase," Froese and Pauly 1997). A similar trend is seen in most of the Food and Agriculture Organization of the United Nations' (FAO) statistical regions of the globe (e.g., both western and eastern Canada; Pauly et al. 2000). These authors have coined an apt term: "fishing down the food web."

THE THREE RATCHETS: "YOU CAN'T GET BACK THERE FROM HERE"

What do we know of the processes that have lead to this sorry state? In fisheries, three ratchet-like processes (Pitcher 2000), continue to contribute to this erosion of biodiversity and ecosystem integrity.

Fishing acts as a selective force on ecosystems by removing long-lived, slow-growing fish in favor of those with higher turnover rates. This process operates both within and among species. When species (or genotypes) become extinct, the past becomes hard to restore, like a ratchet. I term this "Odum's ratchet," recognizing Eugene P. Odum's concerns with human-caused extinctions (for example, see Gibbons and Odum 1993). Natural system maturation processes increasing resilience and complexity of internal trophic and structural relations are essentially the mirror image of unnatural changes caused by chronic exploitation. For example, a cascade of species extinctions characterizes the early stages of the exploitation of ecosystems (Christensen and Pauly 1997, Ward 1997), probably because of the specialized niches of top carnivores.

It is sometimes argued that fishing cannot cause species extinction (e.g., Mace 1997), with a few exceptions among elasmobranchs (e.g., Brander 1981, Casey and Myers 1998). Even when single-species population dynamics parameters that express extinction were invented in the 1950s by Ray Beverton and Sidney Holt,

their relevance was not recognized (Pitcher 1998a). Fortunately, recent work has begun to address these issues more rigorously (Carlton et al. 1999, Musick 1999a, b). For example, Punt (2000) has shown that the fishing mortality that would bring about extinction is only slightly higher than the Maximum Sustainable Yield (MSY) for *k*-selected species. Marine extinctions were reviewed by Malahoff (1997), who quoted two well-known fisheries scientists as averring that most fish are extinction-proof because of high fecundity and planktonic larvae. Even collapsed fish stocks still have millions of fish, and so extinction is unthinkable. In fact, Malahoff points out that local extinctions of concealed sibling species with limited spatial ranges turn out to be widespread. It is important to emphasize that these local extinctions are in fact the critical process in compromising biodiversity, shifting trophic pathways, and altering habitat for other species. Planet-wide extinctions are only the result of the last ever of a series of local extinctions.

The second mechanism, termed "Pauly's ratchet," refers to the psychological tendency for us to relate changes in the system to what things were like at the time of our professional debut: Accounts of former great abundance are discounted as anecdotal, methodologically naïve, or are simply forgotten (Pauly 1995). Sustainability as a policy objective, therefore, tends to be applied to a ratchet-like baseline. Accounts of high abundance are valuable (e.g., Mowat 1984, MacIntyre et al. 1995) because they refer to an early phase of fishing, when a small amount of fishing effort brought about large reductions in biomass.

"Ludwig's ratchet" (termed by M. Healey, *personal communication*, after Ludwig et al. 1993) is the third mechanism, and refers to the generation of additional fishing power through loans that can be repaid only by sustained catches that, on account of stock depletion, can be generated only by further investment in fleet technology. Ludwig's ratchet leads to overcapacity, the single biggest economic problem in fisheries (Mace 1997). Much of the problem has been caused by government subsidies (Milazzo 1998), so it is ironic that that is being addressed by government-funded buy-back programs worldwide, with, however, little apparent success. The catching power of the world fishing fleet continues to expand. Paradoxically, economics was once regarded as the reason why fish stocks cannot be driven to extinction. For example, Beverton and Holt (1955:363) state: "Long before [extinction], declining abundance would cause the fishing intensity to slacken for economic reasons." But it is likely that overcapacity in fisheries drives serial depletion within species (e.g., Orensanz et al. 1998), and, by Odum's ratchet, among species. Moreover, using the Passenger Pigeon and the buffalo as case studies, Farrow (1995) modeled the economics of extinctions, and found that, in a multispecies context, neither privatization, nor price/cost shifts with scarcity were sufficient to avert extinction.

All three of these processes occur automatically, and are ratchet-like in that they proceed in a series of rapid one-way changes, making it difficult to get back to a former state. They can be reversed with varying degrees of difficulty, time, and cost. First, species extinction, Odum's ratchet, is a one-way street (short of Jurassic-park technology), but local extinctions may be reversed by (expensive) reintroductions. Second, reversing Ludwig's ratchet has baffled fisheries economists (Francis 1990, Organisation for Economic Cooperation and Development [OECD] 1997), and will surely involve some draconian instruments. Third, reversing Pauly's ratchet is a cognitive process that requires learning and a shift of a Kuhnian paradigm. Not surprisingly, the general public may find it easier to recapture images of past abundance than fishery scientists. Moreover, indigenous and aboriginal peoples often hold primal abundance in special regard (Haggan 1998a).

Deep-sea fish resources, such as the orange roughy or Patagonian toothfish ("Chilean seabass" in restaurants), are rapidly being subjected to all three of these one-way processes (Gordon et al. 1995). Compared to inshore environments, which in Europe saw a slow development of fishing over 1000 yr, the deep sea is being rapidly depleted worldwide by industrialized "bio-prospecting" (Newman 1995). Sometimes there are almost no regulations on new fisheries, and illegal fishing abounds (Convention for the Conservation of Aquatic Marine Living Resources [CCAMLR] 1999), but despite a trend to management, and single-species quotas for deep-sea fish set by governments and international organizations (Merrett and Headrich 1997), the impact on marine ecosystems of massive reductions in the populations of long-lived, deep-sea fishes has yet to be evaluated, a clear example of a failure to apply the precautionary principle.

Acting in conjunction with these ratchet-like processes, competition for scarce resources stimulates the adoption of new technology, and leads to pathological serial depletion of fish stocks by geographical area and by shifting of target species.

REBUILDING AS A NEW GOAL FOR FISHERIES MANAGEMENT

The most likely scenario for world oceans is that depletion from these ratchet processes will continue to erode both biodiversity and wealth-generating productivity. To reverse the trend, I propose a three-layered agenda. First, we have to establish conceptual goals for the management of fisheries that are firmly rooted in how ecosystems function. Second, we need a fisheries assessment methodology that is capable of addressing this conceptual goal. Third, we need practical everyday tools that can implement remedial actions. This is a tall order, as we need to do no less than reinvent fisheries science to do this. My suggestions are outlined below.

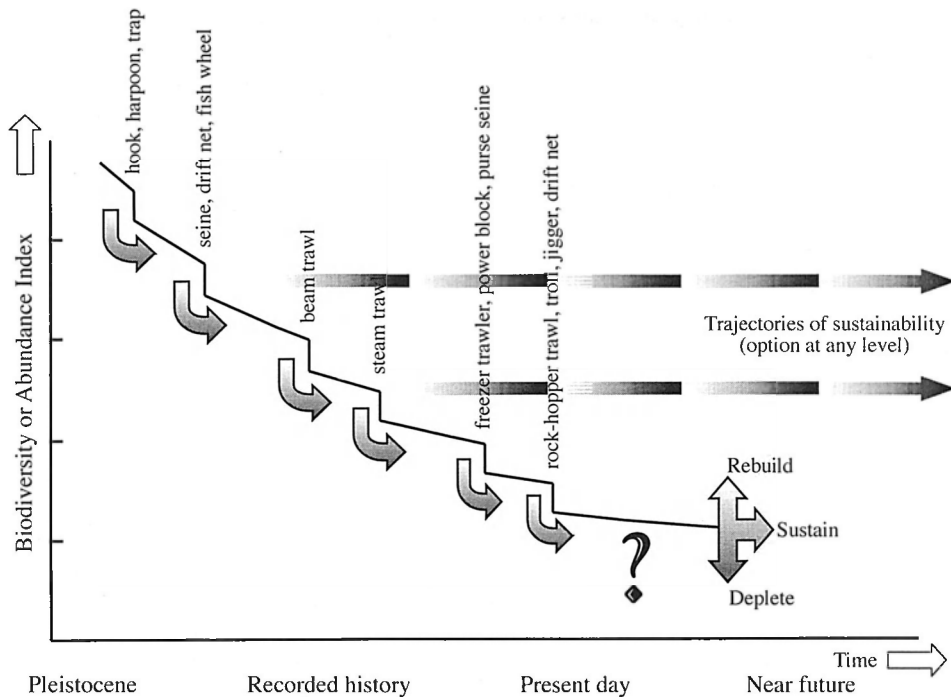


FIG. 1. Schematic representation of the loss of biodiversity in aquatic ecosystems since prehistory. The vertical axis represents total abundance, biomass of top predators, a suitable diversity index, or an ecosystem breadth measure. The horizontal axis is time since the late Pleistocene origin of modern, cooperatively hunting, tool-inventing humans ($\approx 120\,000$ yr BP in Africa). The stepped downward line represents serial depletion of ecosystem resources as new fishing technologies are invented by humans. Some of the principal fishing gear inventions are indicated. Horizontal arrows represent sustainability, which in theory may be pursued at any level of depletion if fisheries assessment, management, and enforcement instruments are effective. Future options are indicated by the three-way arrow (modified from Pitcher and Pauly 1998).

At one time the supreme goal of fishery management, Maximum Sustainable Yield (MSY; Roedel 1975) has repeatedly been called in question (see Larkin 1977, Barber 1988), so that few agencies would acknowledge it as their goal today. Nevertheless, sustainability in one form or another is almost universally employed to define the population criteria that have proliferated to replace it (e.g., Smith 1993, Smith et al. 1993, Drummond and Symes 1996). But the logical problem is that management for sustainability could maintain any given level of depletion of an ecosystem, without considering the benefits of alternative ecosystems (Pitcher et al. 1999). (Since this paper was written, the same point has also been noted by the National Research Council [NRC], 1999.) Pitcher and Pauly (1998) argue that it matters little whether one supports the optimist or the pessimists in judging whether present-day fisheries science can actually achieve sustainability, since sustainability per se is not the right goal.

In Fig. 1, from left to right, the fisheries ecosystem is depleted, biodiversity is reduced, and internal system links lessened. Successive improvements in fishing technology punctuate slow decline with rapid decreases, representing the outcome of the three ratchet-like processes. Given effective management, an ecosystem can be sustained at any level on the way down. But,

with further depletion by excessive fishing, the choice of rebuilding as an alternative goal becomes more pertinent and attractive.

A continuation of our present trend is depressing: ecosystems full of krill, jellyfish, squids, lanternfish, small fast-growing pelagics, and little else (MacIntyre et al. 1995, Parrish 1995, Pitcher and Pauly 1998). These are not the oceans that fisheries scientists have in mind as they diligently work in fisheries management agencies around the world, but, sadly, within less than one hundred years, that is what is likely to result from their efforts. I contend that only a conceptual shift from sustaining to rebuilding stands any chance of preventing this from happening.

In an analysis of the consequences of the rebuilding criterion, Pitcher and Pauly (1998) illustrate that in an emerging era of fish product scarcity (OECD 1997), economic advantage lies in restoring former species and abundance. Table 1 compares the costs and benefits under policies of depletion (effectively, the status quo), sustaining (what most agencies hope is the status quo), and rebuilding (the policy advocated here). The table sets out relative value, total resource value, and costs of harvest, enforcement, monitoring, processing, and employment. The final column evaluates the expected profit, which is maximized under a rebuilding policy.

TABLE 1. Comparison of economic benefits from policies that deplete, sustain, or rebuild the fisheries ecosystems in Fig. 1 (modified from Pitcher and Pauly 1998).

Strategy	Product value	Total resource value	Harvest cost	Enforce-ment cost	Monitoring cost	Processing cost	Employ-ment in sector	Profit
Deplete	low	stable	low	low	low	high	high	low
Sustain	medium	rising	high	high	high	medium	low	medium
Rebuild	high	rising +	low	low	low	low	low	high

Moreover, as prices for traditional “table fish” rise in a future in which fish demand increasingly outstrips supply, jurisdictions that operate a rebuilding policy are likely to outperform economically those with a sustainable policy. Pitcher (2000) outlines extra advantages of ecosystem rebuilding. These include detecting clear upward-trending signals, lessening conflict, clarifying management objectives, benefiting a wider group of resource users, and harnessing diverse groups to monitor resources throughout the ecosystem, such as the public acting as willing sentinels of rebuilding. Nevertheless, as with any change in management, the short-term costs of adjustment will likely act against some resource stakeholders. While the default assumption should be that rebuilding is likely to maximize economic benefit to society, as an exception there may be some fisheries, such as prawns and lobster, where preservation of the status quo would do that. But the “sustainability” paradigm has the disadvantage of deflecting attention from what might be achieved.

The concept of sustainability

While the term “sustainable development” has been questioned (Frazier 1998) because it has been interpreted in a way that fails to acknowledge ecological limits (e.g., ecological footprints; Folke et al. 1998), conservationists hold that sustainability has a broader definition, encompassing social, as well as biodiversity factors; thus, my argument is valid only against a narrow single-species definition of sustainability. Hence, they argue that rebuilding ecosystems is really just a more subtle form of sustainability. Semantically, this is correct, but I contend that it is more helpful to not use the term sustainability in the context of fisheries, where “sustainable social goals,” “sustainable economic goals,” and “sustainable ecological goals” have clearly been mutually incompatible in real management situations, such as in Newfoundland after the collapse of the cod fishery.

A NOVEL METHODOLOGT FOR THE MODEL RECONSTRUCTION OF PAST ECOSYSTEMS: “BACK TO THE FUTURE”

Rebuilding implies reconstructing elements of past ecosystems. A trophic mass-balance model, ECOPATH (Polovina 1984, Christensen and Pauly 1992, 1993), and two derived dynamic simulations, ECOSIM (Walters et al. 1997) and ECOSPACE (Walters et al. 1998, 1999),

can be used to model past ecosystems. Full details of the modeling system are presented in these references, only a brief summary is presented here.

Using data on fisheries catch by sector, production to biomass ratios, consumption rates, and a diet matrix for up to 50 defined components of an ecosystem, ECOPATH tallies the flows of matter within the components of a system, defines trophic levels, and can be used to estimate biomass given diet, mortality, and consumption rates. ECOSIM is capable of evaluating the impact of changes in fishing rates selectively across gear types, and can tune the model to defined time series of biomass estimates. ECOSPACE allows spatial ecosystem modeling by replicating the ECOSIM simulations across a grid of habitat cells: Modeled species groups are allocated habitat preferences. Although complex computationally, these models are conceptually simple, being accounting sheets for diet, biomass, and the import and export of matter (or energy) among predators, prey, grazers, and plants. Biomass, energy, carbon, or nutrients can be used as currency of ecosystem pools. By using the principle of mass-balance, ECOPATH models are grounded in physics (the first law of thermodynamics, and the conservation of energy and matter).

It is an advantage that a preliminary model of an exploited ecosystem can be built without requiring much data, as many variables can be estimated from inputs and outputs by solving a set of simultaneous linear equations. Then, such preliminary models can be improved by adding new detail without disruption. Moreover, workshops set up to construct ecosystem models are valuable for interdisciplinary teams of ecologists, where specialists in particular taxa are forced to understand each others’ concepts, and express fluxes in an agreed common currency. In addition, the ECOPATH software can act as a database for the work of teams of scientists (e.g., Prince William Sound; Okey and Pauly 1998). Finally, the models are quite easy to explain to nonscientists, and are relatively easy to build and validate.

At present, the ECOPATH modeling system has a number of shortcomings. The mass-balance assumption causes difficulties in migratory species, but is relaxed in ECOSIM and ECOSPACE. Behavioral rules that shift diets are not included, larvae and juveniles have to be approximated, and nonstandard trophic pathways, such as the shunt of primary production from large animals to a loop of microbes (Neill 1998), are not

covered. Keystone effects, described by Lawton and Jones (1995), as autogenic or allogenic ecosystem engineering, modulate, through structure, the fluxes from one ecosystem component to another, but such dynamics are not expressed directly in ECOPATH models. Moreover, the tendency in fisheries work, because of convenience, has been to express ecosystem fluxes in terms of biomass, which can be a misleading indicator of a species' role, status, and response to fishing (Grimm 1995). Nevertheless, these shortcomings are being addressed by enhancements to the model algorithms; for example, uncertainty, spatial, and seasonal elements have been addressed in the past two years. The power of ECOPATH, ECOSIM and ECOSPACE, therefore, lies not only in their ability to analyze system-wide responses to fishing, but also in their readiness to be improved.

These techniques have recently been harnessed to a new multidisciplinary methodology for the model reconstruction of past systems. "Back to the Future" (BTF; Pitcher 1998b, Pitcher et al. 1999) employs traditional or local environmental knowledge (TEK or LEK; Hunn 1993), historical documentation, and archaeology (including ancient DNA or molecular archaeology), to facilitate ecological modeling of past systems. Past and present ecosystems, from plankton through fish, marine mammals, and seabirds, are modeled using the ECOPATH and ECOSIM techniques, including simulations that address "what if" questions, such as changes in fishing practices, the closure of areas, or local extinctions of species. Evaluation of a series of such reconstructed ecosystems since ancient times can illustrate dramatically how past marine harvests have progressively foreclosed our future economic options. The BTF methodology supplies a practical direct use for the knowledge of maritime historians, archaeologists, ecological economists, fisheries ecologists, and the TEK of indigenous peoples. It gives all these participants an exciting common goal. The BTF process easily satisfies all but the fifth of the six principles set out by Costanza et al. (1998) for governance of the world's oceans: responsibility, scale-matching, precaution, adaptive management, full cost allocation, and participation. Most alternative methods do not perform as well as this.

Moreover, TEK, if not denied, then a voice often marginalized in conventional fisheries science (Richardson 1992, Kuhn and Duerden 1996; M. S. Weinstein, *unpublished manuscript*), is strengthened in the BTF process by a cross-validation with ecological science (see Johannes 1981, 1987, Johannes et al. 1993). Thus, it may be endowed with a real and valuable role in shaping future fisheries policy (Haggan 1998b). Because it is generally not structured in the same way as ecological science, at first sight it appears difficult to entrain TEK to the BTF process, but through the use of carefully designed questionnaires and interviews, it is possible to break the problem down into simple

choices; for example, presence and absence, place and time. Abundance can be scored relative to other times, or relative to other organisms in the ecosystem (e.g., Salas et al. 1998). This partnership between TEK and ECOPATH can provide a powerful description of past ecosystems (Haggan et al. 1998).

Fig. 2 is a diagrammatic representation illustrating the BTF method (Pitcher et al. 1999). The triangles represent ECOPATH models, which are drawn up for the present day, for a series of specified past times, and for a range of alternative futures. Triangles are used because the ECOPATH model is conventionally represented in the way where the vertex angle and height of the triangle are scaled to biodiversity and internal connectance. The boxes represent species, and the size of boxes represent the relative abundance. (Note that the boxes could equally well represent genetically distinct lineages within a species.) Lines between boxes show the time line of species, ending when local extinction occurs (solid circles). Information about what species were present comes from archaeology, traditional knowledge, or documents represented by the respective symbols. Abundance estimates may be given as trial input to the ECOPATH model, which can then be used to adjust the values to be compatible with the trophic web.

Future ecosystems are represented by alternative models to the right of Fig. 2. They range from heavily depleted ecosystems, filled with low trophic level resources, ranging through maintenance of the present status quo (already depleted, we note), to various ecosystems restored to some semblance of the past. The importance of past ecosystems, once validated, in the policy choice process is that, since they have actually occurred, they must "work." Hence, the "Back to the Future" process can set realistic policy goals for restoration compared to sets of "single species" goals that would otherwise likely be mutually incompatible. "Back to the Future" therefore illustrates how critical ecosystem modeling is for fisheries and ecosystem restoration. The choice of future ecosystem modeling as a policy goal depends on the evaluation of benefits and costs of restoration using ecological economics, and will be the subject of future publications.

The historical times at which it may be appropriate to draw up models will vary among sites. For example, in the Strait of Georgia, British Columbia (Pitcher 1998c), ECOPATH models were constructed for (1) the present day; (2) for 100 yr BP, before the huge modern expansion of salmon fisheries; and (3) 500 yr BP, before contact of native peoples with European settlers and the expansion of the fur trade (Dalsgaard et al. 1998). Biomass of 24 modeled ecosystem components included macrobenthos (Aguilera 1998), capelin, and eulachon (Hay 1998), lingcod (Martell and Wallace 1998), marine mammals (Winship 1998), and fisheries (Wallace 1998). Pitcher (1998d) discussed the possible occurrence of Steller's sea cow and sea otters, while

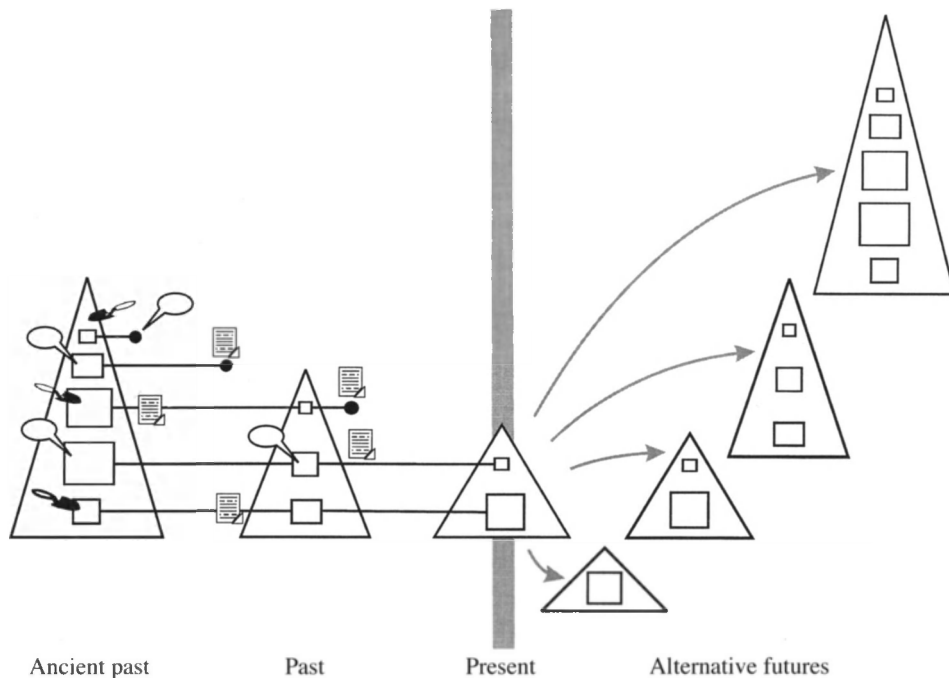


FIG. 2. Diagram illustrating the "Back to the Future" methodology for the evaluation of past ecosystems. Triangles represent three ECOPATH models, constructed at appropriate past times, where vertex angle and height are inversely related to biodiversity and internal connectance. The timing of models depends on the locality, the dawn of quantitative documentary evidence, and major shifts in resource history; a fourth model might be drawn up for the premodern human, late Pleistocene era. Time lines of some representative species in the models are indicated, where the size of boxes represents relative abundance, and solid circles represent local extinctions. Sources of information for constructing and tuning the ECOPATH models are illustrated by the symbols for historical documents (paper), archaeological data (trowel), and the traditional environmental knowledge of indigenous peoples (balloon) (diagram modified from Pitcher et al. 1999). Alternative future ecosystems are drawn to the right.

Williams (1998) analyzed coastal petroglyphs and pictographs for evidence of lost aquatic species (e.g., Crockford 1994), and Danko (1998) surveyed linguistic evidence. At an interdisciplinary workshop (Preikshot et al. 1998), the data were compared with earlier accounts of problems in the Strait's fisheries (Glavin 1996). The scope of the models and conclusions are preliminary and far from definitive, so that it needs a great deal more effort before the work can be credible in informing policy choices for the Strait of Georgia (Pauly et al. 1998b). Nevertheless, it comprises the first published example of the "Back to the Future" process.

In the North Sea, the times chosen to construct ECOPATH models might be: (1) 50 yr BP, before large scale modern fisheries; (2) 200 yr BP, before mechanized fisheries, but after the collapse of the vast Scania herring stock (Cushing 1987); and (3) 1000 yr BP, before the expansion of the Hanseatic league and the rise of modern trade. In the Mediterranean, more ancient model times might be practicable where supported by records of Bronze Age civilizations (e.g., Wachsmann 1998). Where possible, it would also be desirable to draw up a late Pleistocene model prior to the impacts of modern humans, where it may be validated by archaeology. In human-made lakes, where there is no past

ecosystem, an adapted procedure can evaluate alternative future ecosystems (e.g., Lake Nasser; Pitcher 1999).

Toward an aquatic restoration ecology: the need for practical techniques

What practical management instruments for fisheries will allow ecosystem restoration as a policy objective in the face of uncertainty? Here I discuss no-take reserves, and reintroduction of species, both of which have terrestrial analogues.

Restoration ecology in terrestrial environments seems further advanced than its aquatic equivalent (Dobson et al. 1997), in that terrestrial restoration ecology has a powerful set of tools to aid recovery of degraded systems. Terrestrial reserves provide baselines against which to judge human impacts (Arcese and Sinclair 1997), and restoration is also viewed as a necessary hedge against loss from natural causes (Sinclair et al. 1995). Habitat preservation is regarded as the essential template upon which species conservation must be founded: "Habitats can only be preserved if they are treated as a renewable resource; otherwise all habitat will decay to zero" (Sinclair et al. 1995:585). At first sight, it may be difficult to see how this relates

to aquatic species. With major exceptions like coral reefs, rocky shores, and kelp forests, most fish lack a physically tangible habitat made of rooted plant architecture like most terrestrial animals. But the concept needs extending only a little to encompass the trophic structure of the ecosystem. For example, the great marine populations of fish are bounded by tangible ocean structures (Bakun 1996). Both of the practical techniques direct terrestrial analogues.

The establishment of no-take marine reserves (Guénette et al. 1998) is the first essential practical element of marine restoration ecology; the rebuilding of intact systems is a common goal with terrestrial conservation. If some fishing or exploitation is allowed, the purpose is defeated (Ballentine 1991). No-take marine reserves act like a retirement investment. The immediate returns are low, but in the long-term the accumulated interest on natural capital will pay back sustained dividends. For politicians and decision makers, no-take marine reserves are win-win policies: In the long-term, they will ensure that we have a fishing industry that maximizes wealth, jobs, and food, at the same time as conserving habitat, rebuilding the biodiversity of all ocean creatures, and providing recreational opportunities for humans. As natural ecosystems recover in the absence of harvesting, the abundance of large high-value fish species will gradually increase. Such fish and their offspring become available to commercial and recreational fisheries outside of the reserve area (Bohnsack 1993, 1996, Allison et al. 1998). No-take reserves also act as an insurance policy against mistakes by science and management (Clark 1996, Lauck et al. 1998), although my conclusions suggest that marine reserves have a more important function than bet hedging alone.

The first jurisdiction in the world to have the courage to declare large parts of its ocean permanent no-take marine reserves may have the only healthy sustainable fisheries in 50 yr time. Global figures show that seafood demand is increasingly outstripping supply: The fish product markets of the future will place a valuable premium on high-quality, large, traditional table-fish species. No-take marine reserves that are sufficiently large, of the order of 30% of marine areas, will help to ensure those market opportunities.

But, marine reserves cannot be expected to act on their own (e.g., Allison et al. 1998). If fishing effort displaced by the reserve concentrates into the remaining area, or seeks new targets to harvest, advantages will be compromised. Therefore, marine reserves have to be emplaced in conjunction with normal fishery management devices. For nonmigratory stocks, local communities might be encouraged to establish, monitor, and enforce their own reserve areas (Neis 1995, Pinkerton and Weinstein 1995), and the consent and support of local and aboriginal peoples is vital. Aboriginal groups are often suspicious of no-take legislation, as it is seen as a politically motivated allocation device

directed against their traditional harvest rights (Smythe 1995).

A second practical technique is the reintroduction of fish formerly endemic in the area: Terrestrial analogues are reintroducing wolves to national parks, or the suggested re-establishment of bison on the North American plains (Callenbach 1995). Fish should be reintroduced to protected no-take reserves, as those that wander out risk being caught, like reintroduced wolves shot outside Yellowstone Park. But unlike terrestrial restoration ecology, the ultimate purpose of the reintroduction is that their progeny, having established a viable population, are intended to be caught within limits set by stock assessment.

For example, in Hong Kong, lanterns accurately depicting large demersal fish were decorated with ribbons, and carried on poles in procession by coastal peoples for thousands of years (*Wedding Fish Procession*; 1815 oil painting AH1964.002, artist unknown, Hong Kong Museum of Art, *personal communication*). Giant croakers and groupers from the shallow South China Sea were present up to the 1950s, but are now locally extinct (Pitcher et al. 1998). Establishment of artificial reefs within protected areas (Wilson and Cook 1998, Pitcher et al. 2000a, b) may allow the reintroduction to Hong Kong of these valuable fish, currently imported from the coral reefs of adjacent Southeast Asian countries, and often fished with environmentally unfriendly methods (Erdmann and Pet-Soede 1996).

CONCLUSIONS

Many fisheries ecologists call for ecosystem management (e.g., Larkin 1996, Botsford et al. 1997, Policansky and Magnuson 1998, Sherman and Duda 1999), but there have been few clear statements of what its objective should be. Trying to define alternative Optimal Sustainable Yields for each stakeholder results only in confusion. In contrast, a rebuilding and restoration objective for aquatic resources is neither fanciful, preservationist, nor emotive, because the policy produces tangible economic gains in tomorrow's markets. The same goal is now well established in terrestrial ecology (Sinclair et al. 1995), and a similar approach is advocated to solve the problems of forestry (Marchak 1995), which in many respects parallel those of fisheries. Restored systems have higher biodiversity, allowing more market niches for seafood products. Moreover, restored marine ecosystems maximize benefit to society because they have higher value from multiple users, satisfying conservation objectives, at the same time as providing wealth and reducing allocation conflicts.

In economic terms, natural ecosystems produce both goods and services. Fisheries represent most of the goods. But Costanza et al. (1997) estimate that the economic value of ecosystem services is 63% of the total global value of \$33 trillion. Half of this derives from coastal ecosystems that provide services esti-

mated at \$400 000 per km². Less than 5% is from fishing. The rest is the estimated service value to humans of nutrient recycling, disturbance regulation, recreation, cultural use, raw materials, habitat refugia, and biological control. We can expect restored ecosystems to have higher service values.

Unlike sustainability, which needs to be carefully defined because economic and ecological sustainability easily find themselves at odds, a management goal for rebuilding aquatic systems brings a number of benefits. This goal can be clearly associated with greater economic, social, and ecological status, and so can easily be understood and adopted by a nonscientific public. Pitcher (2000) argues that this benefit might be lost if decision making is subverted by commercial or industrial interests, or by pressure not to inconvenience the politicians of the day. Dayton (1998) calls for a "reversal of the burden of proof" to oblige fisheries to demonstrate that their activities cause no ecologically significant long-term changes. Additionally, rebuilding should not foreclose future options to profit from the seas, to recover the lost opportunity costs of collapsed resources, and to maximize the value of services provided by intact aquatic ecosystems. Gro Brundtland (1997) is optimistic that "politics that disregard science and knowledge will not stand the test of time," but time is what we do not have a lot of for the world's depleted oceans.

ACKNOWLEDGMENTS

I thank Pam Brown, Johanne Dalsgaard, Sylvie Gu  nette, Nigel Haggan, Trevor Hutton, Tom Okey, and Daniel Pauly for commenting on the manuscript, and Tony Sinclair, Rashid Sumaila, and Carl Walters for discussion of ideas. The material in this paper was first presented at the Seventh International Congress of Ecology in Florence, July 1998, and I am grateful to Klemens Ekschmitt for the invitation to speak, and thank him, and a total of six anonymous referees, for comments that have improved the manuscript.

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