

TRENDS IN GLOBAL MARINE FISHERIES – A CRITICAL VIEW¹

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Abstract

Individual fisheries are generally perceived as one fleet exploiting one or several target species. Their welfare then depends on the relation between the size of each fishery relative to the size of population(s) it exploits, which ‘stock assessments’ are supposed to evaluate and ‘fisheries management’ is supposed to adjust. The vision of fisheries that will be presented here, however, is that of a global system spanning all oceans. It is the result of an expansion which began in Europe, North America and Japan over a century ago, and which is now being completed by largely uncontrolled industrial fleets operating in the deepest waters of the Southern Oceans.

The major consumer countries, mainly the EU, the United States and Japan, are largely unaffected by the local depletions these fleets induce, in spite of globally declining catches, buffered as they are by an integrated global market that causes a large and increasing fraction of world fish catches to be consumed in countries other than those where the catches are being made. This global fishery, fed by enormous subsidies in developed countries and ‘subsidized’ by the need for foreign exchange in developing countries, has

a substantial impact on marine ecosystems, with the biomasses of large and mid-sized target species and associated organisms generally reduced by one order of magnitude a few decades after a fishery opens. This effect can be detected by declining trends of the mean trophic level of fisheries landings, a process now known as ‘fishing down food webs’. We briefly review, in the context of Large Marine Ecosystems (LMEs), the work documenting this global phenomenon, which implies that fisheries gradually ‘spread’ from a few targeted species contributing the bulk of the catch to a situation where essentially all palatable taxa are targeted, with each taxon (‘diversity’) contributing a small part (‘evenness’) of the total catch. This is demonstrated through the application of an indicator (‘BA-DAP index’), derived from work by R.M. Warwick, applied here for the first time to the time series of LME catch data for 1950 to 2004. It can be anticipated that the form of interaction with marine megafauna and their supportive ecosystems demonstrated here will lead in the next decades to a succession of local extirpation, followed potentially by global extinctions, just as early hunters exterminated the megafauna of newly accessed continents or

1. This chapter is updated from material presented at the 5th ICEF: Environmental Future of Aquatic Ecosystems Conference, Zürich, 23–29 March 2003; and Pauly *et al.* (2008).

islands. Confronting the ecosystem impact of this exploitation system will require a new mode of thinking on how humans and marine wildlife can co-exist on Earth.

Yes, we can and do exterminate our prey

Notwithstanding humans' alleged 'biophilia' (Wilson 1993), it is now well established that hunters, given the means and the opportunity, will exterminate the animals they hunt. Understanding the underlying processes is important, as they provide a framework for understanding the war of extermination presently waged against smaller and mid-size mammals in Africa ('bushmeat'; Bowen-Jones 1998) and against large fishes in the world ocean, the latter process being commonly known as 'fishing'.

Perhaps the best studied, and most illustrative of these wars of extermination, was conducted 13,000–12,000 years ago by 'Clovis' hunters in North America, so named after the site where the first of their magnificent fluted arrow and spear points were found. Notwithstanding a long tradition stating the contrary, the Clovis hunters were probably not the 'First Americans', these probably having been coastal people, who may have relied on fishing for their subsistence (Dalton 2003). The Clovis people, on the other hand, were apparently the first to tackle the large mammals of the interior. Both archaeology and model studies confirm that their expansion and population growth, and their decimation of 30 species of large and slow-reproducing mammals

of North America (mastodon, giant ground sloth, giant armadillo, western camels, etc.), proceeded in the form of a wave lasting from 800 to 1,600 years (Alroy 2001). Given the human life span, and the difficulty of all societies, but especially of preliterate ones, to convey quantitative information on past animal abundances across generations (Pauly 1995), this time span was sufficient for the Clovis hunters living past the crest of this wave to fail to realize what their ancestors had done and lost (Alroy 2001).

The full realization and understanding of this event is prevented by those who, contrary to the rather strong evidence provided by a multitude of Clovis points embedded in fossil bones, still argue that this megafauna extinction was driven by climate change. Can we truly deny events which happened to coincide precisely with the arrival of the Clovis hunters and which are supposed to have eliminated, in a few centuries, species that had endured millions of years of environmental change, including glaciations that entombed North America under two kilometers of ice? We shall return to this theme of denial, as it also precludes informed debate about the impact of industrial fisheries.

There is good evidence of a similar hecatomb about 46,000 years ago in Australia, in this case associated with the very first arrival of *Homo sapiens*, and which exterminated the larger representatives of the marsupial fauna that had evolved on that continent, again over millions of years (Roberts *et al.* 2001). Need we stress that there are, here as well, those who say it is some environmental fluctuation that is to blame?

Then there is the extermination of the large, ostrich-like moa in what is now New Zealand, by Polynesians, the ancestors of the present day Maori, who arrived in the late 13th Century, and who took approximately 100 years to exterminate 11 species that required millions of years to evolve (Holdaway and Jacomb 2000). In this case, it seems, few are claiming it is the ‘environment’ that did it.

In Africa, on the other hand, where humans co-evolved with large mammals, the latter survived until recently, by being wary of bipedal primates carrying sticks. Indeed, it is only recently that guns, globalization and human population growth have combined to disrupt this evolved balance, and that the hunting of mammals of all sizes, and the local and international trade of ‘bushmeat’ are now emptying African terrestrial ecosystems of all but the smallest rodents and insectivores (Bowen-Jones 1998, Brashares *et al.* 2004). Here again, the events are too close and easy to trace to their causes for the ‘environment’ to be blamed.

Why all this in a contribution devoted to fisheries, i.e., the killing of fish from waters, rather than mammals on land? One reason is obviously to pre-empt ultimately sterile debates about the recent, massive biomass declines in various large marine ecosystems (see e.g. Christensen *et al.* 2003 for the North Atlantic) being due to anything but overfishing (see also Jackson *et al.* 2001 and Worm *et al.* 2006). Thus, we shall ignore those who deny that fishing impacts on the underlying resources, and that observation of low fish biomass are due either to scientists being

incompetent at detecting or catching the fish in question (as in the ‘Trawlgate’ case described by Malakoff 2002), or the fish having moved elsewhere (Bigot 2002), or because of some regime shift (Steele 1998).

The other reason for using hunting to frame a discussion of world fisheries is to suggest that the very approach we use to define fisheries may in fact be misleading. We are used to discussing fisheries in terms of their target species, e.g., the ‘North Sea cod fishery’, or the ‘Greenland shrimp fishery’, but these labels, which seemingly package distinct, separable set of attributes, are as misleading as identifying giant armadillo, or mastodon hunts by the Clovis hunters. Fact is, these hunters killed all large, slow moving targets they encountered, and ended up, once these were gone, with a species assemblage composed mainly of small mammals (with the large bison, moose and bears among the few exceptions), and whose components were much harder to get close to, and less rewarding to hunt. In fact, given these hunting patterns, Alroy (2001) found that in his simulations, “the only way to prevent a size-selective mass extinction is to assume that humans strongly prefer to hunt small game”.

Similarly, when fishing starts in a new area, it is the large fishes that go first, given that they can be readily caught in the first place. Large fishes are relatively easier to catch than small fishes, and tend to provide a better return on energy expended, be it in form of muscle power (rowing boats, reeling hooked fish), or in terms of financial return on fuel costs (large fish usually fetch higher prices than an equivalent weight of small fish, the

latter being often used only for fishmeal, when not discarded, Pauly *et al.* 2002). Large fish, with their low natural mortalities and relatively high age at first maturity (Pauly 1980, Froese and Binohlan 2000), cannot sustain much fishing pressure and they decline (Denney *et al.* 2002, Cheung *et al.* 2007), forcing the fishers to either move on to smaller fishes, and/or to other, previously unexploited areas. As larger fish tend to have higher trophic levels than smaller fish – indeed, the latter are usually the prey of the former – this process, now called ‘fishing down marine food webs’, invariably leads to declining trends in the mean trophic level of fisheries landings (Pauly *et al.* 1998a).

This suggestion was initially challenged (Caddy *et al.* 1998), but subsequent work, summarized in Table 1, shows the ‘fishing down’ phenomenon to be ubiquitous. For the various countries/areas covered in that table, ‘fishing down’ implies a transition from fisheries targeting large fish (e.g. northern cod) to fisheries targeting smaller fishes (e.g. capelin), or invertebrates (e.g., deep water prawn [*Pandalus borealis*] and crab [*Chionoceters opilio*]). However, the broad pattern is that of the fishers in a country or area targeting a succession of species (just as the Clovis hunter did), until the residual species mix ceases to support a fishing (or hunting) economy (unless artificially maintained via subsidies, Sumaila *et al.* 2007).

Table 1. Contributions demonstrating the occurrence of ‘fishing down marine food webs’ using local/detailed datasets, following the original presentation of this phenomenon by Pauly *et al.* (1998a), based on the global FAO landings dataset.

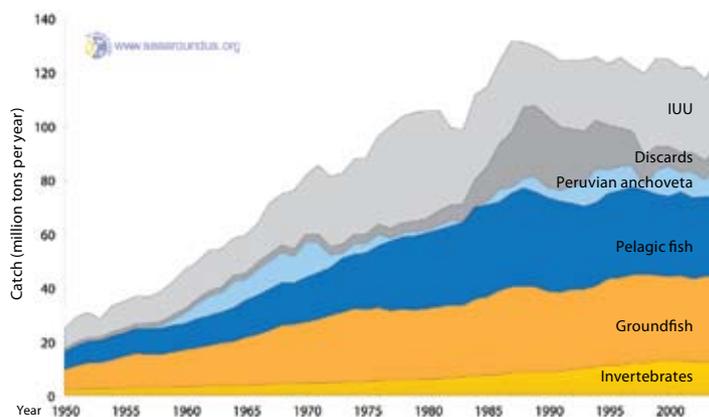
Country/area	Years	Decline	Source and remarks
Iceland	1900–1999	1918–1999	Valtysson and Pauly (2003), based on comprehensive catch database of Valtysson (2001).
Celtic Sea	1945–2000	1946–2000	Pinnegar <i>et al.</i> (2002), based on trophic levels estimated from stable isotopes of Nitrogen.
India	1950–2000	1970–2000	Bhathal (2005), Bhathal and Pauly (2008).
Gulf of Thailand	1963–1982; 1963–1997	1965–1982; 1965–1997	Christensen (1998); Pauly and Chuenpagdee (2003).
Eastern Canada	1950–1997	1957–1997	Pauly <i>et al.</i> (2001), based on data submitted to FAO by Fisheries and Oceans Canada (DFO).
Western Canada	1873–1996	1910–1996	Pauly <i>et al.</i> (2001), based on comprehensive dataset assembled by Wallace (1999).
Cuban EEZ	1960–1995	1960–1995	Pauly <i>et al.</i> (1998), Baisre (2000).
East Coast, USA	1950–2000	1950–2000	Chuenpagdee <i>et al.</i> (2003); emphasis on Chesapeake Bay.
Chinese EEZ	1950–1998	1970–1998	Pang and Pauly (2001).
West Central Atlantic	1950–2000	1950–2000	Pauly and Palomares (2005), based on FAO data (Statistical Area 41, see Ackefors this volume), disaggregated into USA (North) and other countries (South).
World, all fishes	1950–2000	1950–2000	Pauly and Watson (2003), based on spatially disaggregated data.

It is our contention that, given a continuation of effective fishing effort increases driven by the provision of damaging subsidies (Clark and Munro 2002, Clark *et al.* 2005, Sumaila *et al.* 2007) in the face of globally declining catches (Watson and Pauly 2001), and in the absence of a global network of relatively large marine reserves (Russ and Zeller 2003, Wood *et al.* 2008), similar in scope to the national parks that now protect the large mammals of Africa, large marine fishes, both pelagic (some tuna, many sharks and billfishes), and demersal (groupers, large croakers, etc.) may likely be fished into extinction in the next decades. Sadovy and Cheung (2003), and Dulvy *et al.* (2003) provide strong support for this view. The fisheries targeting these species will obviously go bankrupt in the process, though they may continue to last for a while if propped up by sufficient subsidies (Sumaila *et al.* 2007).

Globally, marine fisheries catches, as reported by countries to the United Nations Food and

Agriculture Organization (FAO), increased from less than 20 million t·year⁻¹ in the early 1950s to a peak around 80 million t·year⁻¹ in the mid 1990s (Figure 1, adjusted for over-reporting by China, see Watson and Pauly 2001). Important, however, is that these numbers are only 'reported' landings, and do not include catches taken by fishing gears, but not landed and used (i.e., discards), nor do these figures address Illegal, Unreported and Unregulated fisheries sectors (IUU). Especially unreported and unregulated sectors can be significant, and contribute, particularly in the form of developing countries' small-scale subsistence fisheries, often far more to total catches than the commercial catches these countries officially account for in their national and international (FAO) reporting (e.g Zeller *et al.* 2007, Zeller and Pauly 2007). Considering current best estimates of these components, suggests that total global marine fisheries catches likely peaked around 120 million t·year⁻¹ (see Figure 1).

Figure 1. Global fisheries catches by major groupings (invertebrates, groundfish, pelagic fish and Peruvian anchoveta [*Engraulis ringens*]) adjusted for over-reporting by China (Watson and Pauly 2001), and estimates of global discards (based on Zeller and Pauly 2005) and IUU catches (Illegal, Unreported and Unregulated, based on Pauly *et al.* 2002).



More reasons for pessimism

The pessimistic outlook for fisheries outlined above has antecedents, notably in Ludwig *et al.* (1993), who pointed at the ease with which, in the absence of a clear experimental context, field data from collapsed fisheries can forever be contested by the fishing industry, a point developed further by Rosenberg (2003), and which, given the capture of management bodies by that same industry (see e.g., Okey 2003), leads to paralysis at best, and to continued erosion of the resource base at worst. This problem, which is very real in developed countries, many of which pride themselves of the way they incorporated science-based advice into policy making, appears largely insurmountable in developing countries, where science is a fragile import. In these countries, the great needs for food and incomes by large human populations, which should lead to more careful use of natural resources, in fact provide decision-maker with a ready excuse for what is, in fact, the wholesale destruction of food production systems.

Indeed, exports have become a major issue in fisheries, with marine products being amongst the most heavily traded commodities (Pauly *et al.* 2002, Alder and Sumaila 2004). The general trend is that the shortfall of products from traditional fishing grounds in the EEZ of developed countries is being compensated for by exploitation (often via distant water fleets) of developing countries (Pauly and Watson 2003). This implies, given the 'debts' that most developing countries have run with respect to international lenders, that marine resources and their underlying eco-

systems suffer from increased pressure. Examples are provided by the countries of West Africa, whose dependence on financial support from the European Union forces them to agree to fisheries agreements providing access to European fishing fleets under terms that appear rather unfair to these countries (Kaczynski and Fluharty 2002). Similar developments appear to be underway off East Africa (Jacquet and Zeller 2007a, 2007b). Another example is Argentina, whose demersal resources, in the early 1980s among the few that were both large and underexploited, have now collapsed under the pressure of both national and international fleets, licensed for their ability to generate foreign exchange (Sánchez 2002).

Also important is the absence of international institutions (i.e., set of rules) capable of action that would reverse, or at least halt, in international waters, trends such as presented above, and impacting on large pelagic fishes (see e.g., Pauly and Maclean 2003 for the case of the North Atlantic, where such institutions and organizations had a long time to develop, but didn't). An exception could possibly be the World Trade Organization (WTO), as it is the only international body with the ability to enforce its rules. As of this writing, WTO members are considering proposals to reduce or eventually even eliminate subsidies for fisheries (Sumaila *et al.* 2007). The current, 'Doha round' of negotiations may fail, but the issue will remain until it is resolved.

Parrish (1995, 1998) developed rather pessimistic scenarios for the future of fisheries, arguing that, given present trends and pressures,

and their apparent irreversibility, it is mainly small, non-palatable, non-schooling fish species that will be able to survive the present onslaught. Conversely, he argued that the species that fisheries presently target (Table 2) will not survive what he called the “late Holocene”, and which many others refer to as the 6th Extinction (see e.g. Eldredge 2001).

Table 2. Marine fish that are unlikely to survive, given continuation of present fisheries trends (adapted from Parrish 1995, 1998, and Pauly 2000).

Major features	Representative groups
Large- to moderate-sized, predaceous, territorial reef fishes and rockfishes with late age at maturity, very low natural mortality rates and low recruitment rates vs. adult stock size.	Snappers, sea basses, emperors, rockfishes, sea breams.
Large- to moderate-sized shelf dwelling, soft bottom predators susceptible to bottom trawling.	Cods, flounders, soles, rockfishes, croakers, skates.
Large- to moderate-sized schooling midwater fishes susceptible to midwater trawling.	Hakes, rockfishes, armorheads, rougheyes.
Large- to moderate-sized shelf dwelling, schooling, pelagic fishes.	Bonitos, sierras, capelin, eulachon, salmon, sharks.
Any species with exceptionally high monetary value.	Bluefin tuna, red snappers, halibuts, medicinal fishes, aquarium fishes, groupers, salmon, red mullets, billfishes.

Whether or not Parrish’ scenarios will come to pass, there will be a need to monitor the impact of fisheries on their supporting ecosystems, a task for which appropriate indicators are required. Indicators, beyond the catches themselves, which can be used for this purpose include:

1. The taxonomic resolution of the reported catch over time (which however, has an inherent, if unknown bias, see Figure 2);
2. The Marine Trophic Index (MTI), i.e. mean trophic level of the catch (see Table 1 and text below), and the related Fishing-in-Balance (FiB) index (Pauly *et al.* 2000); and
3. Stock-Catch Status Plots, based on catch time series (Pauly *et al.* 2008).

These indices are relatively easy to apply, as they require only catch time series (e.g., in the form of the global FAO catch dataset; see below) and estimates of trophic levels, available for essentially all fish species in the world (from FishBase; Froese and Pauly 2000) and for all commercially exploited groups (from the database of the *Sea Around Us* Project; see www.seaaroundus.org). These indices are here applied to Large Marine Ecosystems (LMEs), of which 64 are now defined (Sherman and Hempel *in press*)², and which are gradually emerging as convenient entities to report fisheries trends (Pauly *et al.* *in press*).

The ‘Biomass-Abundance-Difference in Area in Percent’ index (BA-DAP) – an indicator based on the taxonomic resolution of the reported catch

As mentioned above, fishers, like hunters, tend to concentrate preferably on large organisms. Thus, their catch initially consists of few species, with even fewer individuals contributing the bulk of that catch. As the initial target species become scarce, more species are targeted and contribute

2. The currently defined 64 LMEs are likely to be re-defined into 66 LMEs through splitting of the previously large and poorly defined Arctic LME, see www.seaaroundus.org.

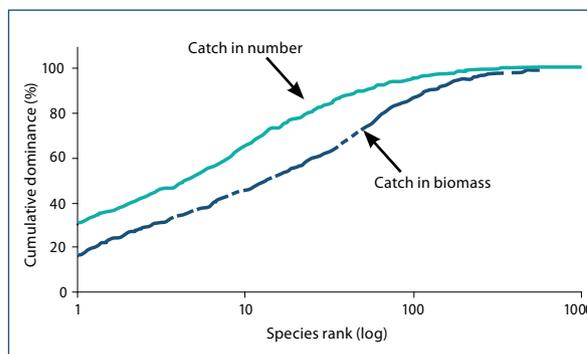
to the overall catch, until at the end, the fisheries ‘spreads’ to all catchable and marketable components of the ecosystem, with none contributing much, the catch having become taxonomically ‘even’. Given such trends, an index is needed which captures the elements of this ‘spread’: i.e. the increasing dominance of catch by species that are large in total number but small in individual biomass (i.e. catch ‘evenness’).

Moreover, the index must be normalized such that it can be used to compare changes in areas with different species richness (and/or with statistical reporting systems considering different levels of taxonomic aggregation).

Here, we propose a *Biomass-Abundance-Difference in Area in Percent (BA-DAP)* index to express the changing contribution of catch from the exploited species. Essentially, Warwick (1986) developed the Abundance-Biomass Curve that presents community structure data in the form of *k*-dominance curves, and consists of a logarithmic x-axis with the ranks of species by declining

abundance and biomass, and a y-axis along which the cumulative percent abundance and biomass by species are plotted (Figure 2). An ‘unstressed’ community is generally dominated by species that are large in biomass but small in abundance (i.e. the biomass curve is above the abundance curve). Conversely, in a ‘stressed’ community, species with small biomass and high abundance become dominant, i.e. the biomass curve is under the abundance curve (Warwick 1986). Similarly, in the context of catch composition, a fishery may initially be dominated by high biomass (large) fish that are relatively low in number. As the fishery develops, catch may be increasingly dominated by species with high abundance but smaller biomass (e.g. small pelagic fishes). The relative position of the abundance and biomass curves can be expressed by a variant of the DAP index originally developed by McManus and Pauly (1990) (here termed the ‘Biomass-Abundance-DAP’ or ‘BA-DAP’ index), calculated from the difference in areas between the two curves, or:

Figure 2. Example of an Abundance-Biomass Curve based on Warwick (1986) presenting community structure data in the form of *k*-dominance curves, here adapted to fisheries catch data for a single year. These curves consist of a logarithmic x-axis with the ranks of species by declining abundance or biomass, and a y-axis with the cumulative percent abundance and biomass by species. The resultant BA-DAP index as presented here (modified from McManus and Pauly 1990) is calculated by comparing the areas underneath the two curves for each year, standardized by the number of taxa reported in that year, as $BA-DAP = (\text{Biomass curve area} - \text{abundance curve area}) / \log_e \text{ number of species}$. It is worth noting, however, that, especially with regards to fisheries catch time series, this approach and the resultant index is sensitive to changes in taxonomic reporting quality, which is a well known historical problem with catch data, and results in currently untested biases.



$$Area = \sum_{i=1}^{s-1} [Ci + (0.5 \cdot Y_{i+1})] \cdot [\ln(i+1) - \ln(i)]$$

$$BA - DAP = (Area_{Biomass} - Area_{Abundance}) / \ln s$$

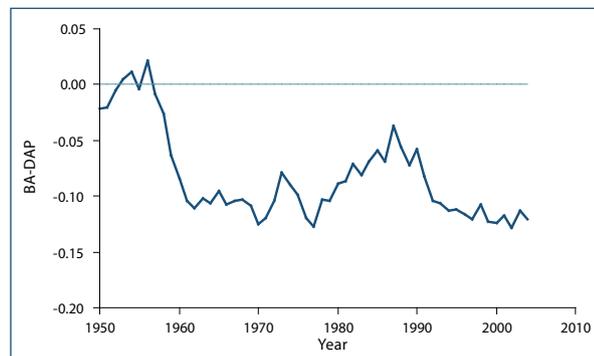
where C and Y is the cumulative and individual percentage of abundance or biomass of the species at abundance or biomass rank i , respectively; s is the number of reported taxonomic groups (species, genus, family or grouping). Thus, the BA-DAP index scales between -1 and +1, with negative values indicating an abundance dominated system.

The BA-DAP index was calculated for each year from 1950 to 2004 for each LME. As the index is normalized by the number of taxa, the index can be readily compared between different LMEs (or other spatial entities) and time. Since catch records by the number of individuals (C) is not available for the majority of fisheries, we obtained a conservative estimate of C by assuming a power relationship between body length

(using maximum size data from the *Sea Around Us* Project and FishBase databases) and weight of each species. This, in practise, amounted to dividing the catch (tonnage) by the maximum length (in centimetres) raised to the third power. To avoid the large number of tiny invertebrates such as *Acetes* shrimp being caught in some LMEs from over-dominating the catch time-series, species with maximum body length of less than 10 centimetres were not included in the analysis.

Figure 3 shows an application by the *Sea Around Us* Project of the BA-DAP index to the catch statistics assembled for 53 of the world's 64 defined LMEs³, based mainly on data submitted to the FAO by member countries (Watson *et al.* 2004). The BA-DAP declined sharply from the mid-1950s to the early 1960s, indicating that the catch from the 53 LMEs quickly became dominated by small-bodied species. In 1950, Atlantic cod (*Gadus morhua*) contributed the largest biomass of catch, but less than 0.01 percent of individuals. By 1962, the catch (in weight) of

Figure 3. BA-DAP index (Biomass-Abundance-Difference in Area by Percent) based on fisheries catches (species <10 cm maximum length excluded) for 53 of the world's 64 currently defined Large Marine Ecosystems (LME) combined. Eleven LMEs did not have sufficient data to calculate the index and were excluded here. A negative index implies abundance dominance, while a positive value suggests biomass dominance. Index modified from McManus and Pauly (1990).



3. The LMEs not included here are the data poor Arctic and Antarctic LMEs.

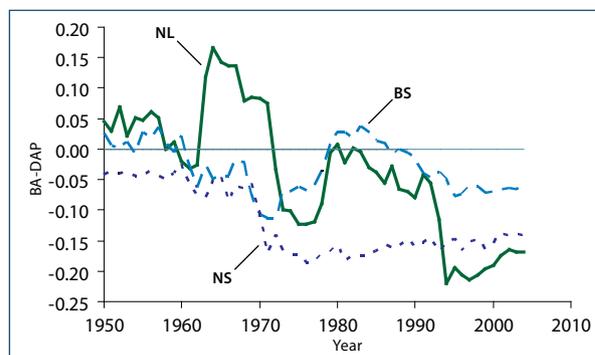
Peruvian anchoveta (*Engraulis ringens*) became twice that catch of cod and contributed 11 percent of the catch in individuals. The BA-DAP index increased moderately in the 1980s but reduced again after 1990. The large increase in Alaska Pollack (*Theragra chalcogramma*) increased the dominance of species with larger body weight between 1980 and 1990. However, the decline of Pollack catches contributed to the subsequent fall of the index. Overall, the average decline in the BA-DAP index can be interpreted as an indication of the increasing ‘stress’ in the reported fisheries catches of the 53 LMEs.

Here, and in the subsequent two sections, we present results for three LMEs as examples, i.e. the Newfoundland-Labrador Shelf LME, the North Sea LME and the Baltic Sea LME.

The Newfoundland-Labrador Shelf LME extends a considerable distance off the eastern coast of Canada, encompassing the areas of the Labrador Current and the Grand Banks. As in some other LMEs, overexploitation is the princi-

pal driver of changes within this LME (Sherman and Hempel in press), although fluctuations in the ocean climate have also been implicated. A description of the changing conditions of the fish and fisheries of this LME is given in Rice (2002). Commercially exploited fish species in this LME include Atlantic cod, haddock (*Melanogrammus aeglefinus*), American plaice (*Hippoglossoides platessoides*), redfish (*Sebastes* spp.), and more recently, snow crab (*Chionoecetes opilio*) and deep water prawn (*Pandalus borealis*). The BA-DAP index for this LME indicates a virtually continuous decline over the 50+ year time period, with two brief reversals in the second half of the 1960s and early 1980s, driven by temporary increases in Atlantic cod catches (Figure 4). The complete collapse of the cod stocks in the early 1990s precipitated another strong decline in the index. Overall, the index illustrates a clear move from a biomass dominated (positive BA-DAP) to an abundance dominated (negative BA-DAP) fisheries system (Figure 4).

Figure 4. BA-DAP index (Biomass-Abundance-Difference in Area by Percent) based on fisheries catches (species <10 cm maximum length excluded) for three example Large Marine Ecosystems (LME): Newfoundland-Labrador Shelf LME (NL), North Sea LME (NS), and Baltic Sea LME (BS). A negative index implies abundance dominance, while a positive value suggests biomass dominance. Index modified from McManus and Pauly (1990).



The North Sea LME is relatively shallow, and is situated on the continental shelf of north-western Europe. The LME includes one of the most diverse coastal regions in the world, with a great variety of habitats (fjords, estuaries, deltas, banks, beaches, sandbanks and mudflats, marshes, rocks and islands). Among its many river systems and estuaries are the Thames, Rhine, Elbe and Ems. A temperate climate and four seasons characterize this LME, with climate being an important driving force (after fishing) of biomass change in the LME. Great Britain, Norway, Sweden, Denmark, Germany, the Netherlands, Belgium and France are the countries bordering the North Sea (Sherman and Hempel 2008). Fishing is a long-established activity in the North Sea LME and there is a wealth of fisheries data (Froese and Pauly 2003). The most important species for human consumption represented in the catch are Atlantic cod, saithe (*Pollachius virens*), Atlantic herring (*Clupea harengus*), European sprat (*Sprattus sprattus*) and flatfishes. Landings from the industrial fishmeal reduction fishery consist mainly of sandlances (sandeels *Ammodytes* spp.) Norway pout (*Trisopterus esmarkii*) and sprat. There are several commercially important shellfish species of molluscs and crustaceans. The BA-DAP for the North Sea LME indicates a steady decline in indicator in the early years, followed by a sudden, steep decline in 1970 towards strongly abundance dominated fisheries (Figure 4). The index has remained steady ever since. It is worth noting that throughout the time period considered here (1950–2004), the index was negative, suggesting that the fisheries in this LME

have been abundance dominated for the entire time period, as illustrated by the predominance of relatively small, yet abundant species such as, e.g., herring (*Clupea harengus*) and sandlances (*Ammodytes*).

The Baltic Sea LME is the world's largest brackish water body, and its catchment area is four times larger than its surface area (Sherman and Hempel 2008), nearly 93 percent of which belongs to nine riparian countries: Denmark, Estonia, Finland, Germany, Latvia, Lithuania, Poland, Russia and Sweden. The non-coastal countries in the catchment area include Belarus, the Czech Republic, Slovakia and Ukraine. Atlantic cod, herring and European sprat dominate the fish community as well as catches in terms of numbers and biomass. The Baltic Sea LME shows an interesting BA-DAP pattern (Figure 4). It initially was marginally positive, suggesting a slight biomass dominance in catches, before declining to an abundance dominated catch pattern until the early 1970s (negative BA-DAP, Figure 4). This was followed by a return to a positive, biomass dominance in the early 1980s, and a subsequent steady decline to an abundance dominance pattern which is maintained today (Figure 4). The decline in index in the 1960s and early 1970s was driven by the increase in catches of herring and sprat, while the 1980 biomass dominance was the result of a brief resurgence of cod catches, with a concomitant substantial decline in sprat catches. The subsequent collapse of the cod stocks and resurgence of sprat catches resulted in the most recent decline in the index (Figure 4). The predominance of three taxa in the Baltic Sea catches,

namely herring, European sprat (both abundance dominant) versus cod (biomass dominant), clearly illustrates the response behaviour of this new index.

This illustrates the versatility of this new BA-DAP index, which integrates the change in contribution of each species in the catches and the 'stress' level of the faunal community (as expressed in reported catches) in a single, standardized manner, and illustrates in yet another way the effects of fishing on marine ecosystems.

The Marine Trophic Index and the FiB index

Pauly *et al.* (1998a) identified a worldwide decline in the trophic level of fish landings by assigning a trophic level to each species in the FAO landings data. This pattern, now known as 'fishing down marine food webs', has been shown to be relatively wide-spread, especially when investigated on a smaller scale (Table 1). The ubiquity of 'fishing down marine food webs' is one of the reasons why the Convention on Biological Diversity (CBD) adopted the mean trophic level of fisheries catch, renamed as *Marine Trophic Index* (MTI) as one of eight biodiversity indicators for "immediate testing" (CBD 2004, Pauly and Watson 2005).

However, diagnosing 'fishing down' from the mean trophic level of landings is problematic, as landings only crudely reflect abundances. Furthermore, as fisheries overexploit their local resource bases, e.g. in inshore waters, they will tend to move further offshore to outer shelf waters and beyond (Morato *et al.* 2006). During this

spatial expansion, fisheries access previously unexploited stocks, resulting in the MTI calculated for the whole shelf, which may have declined at first, increasing again, especially if the 'new' landings are high. Thus, at the scale of an LME, a trend reversal of the MTI may occur when fisheries expand geographically. This is the reason why the diagnosis as to whether fishing down occurs or not, now performed for many LMEs (Sherman and Hempel in 2008), generally depends on the species composition of the landings, which may indicate whether a geographic expansion of the fishery has taken place.

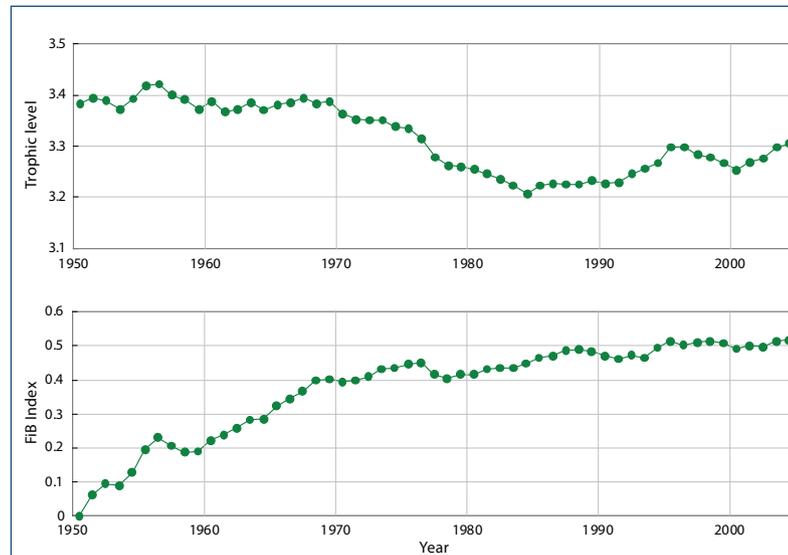
To facilitate such evaluations, a *Fishing-in-Balance* (FiB) index has been developed (Pauly *et al.* 2000). This index is defined such that its value remains the same when a downward trend in mean trophic level is compensated for by an increase in the volume of 'catch'. This is based on the pyramidal nature of ecosystems and a average transfer efficiency of ten percent between trophic levels (Pauly *et al.* 2000). Obviously, the FiB index will decline when both the MTI and landings decline, which is to be the case in many LMEs (Sherman and Hempel 2008). In contrast, the FiB index will increase if landings increases more than compensate for a declining MTI. In such cases (and also when landings increase and the MTI is stable or increases), a geographic expansion of the fishery has likely taken place, i.e. another part of an ecosystem is being exploited (Bhathal and Pauly 2008). This index has been designed such that its absolute value is of no concern, i.e. that the change of the index can be assessed from any baseline. Generally, it is stand-

ardized to have a value of zero in 1950.

Figure 5 (based on Pauly *et al.* 2008) presents the MTI and FiB index for 53 LMEs combined, but with two groups of fishes excluded: Peruvian anchoveta (*Engraulis ringens*) and large pelagic fishes (large tunas and billfishes). The very localized South American fishery for Peruvian anchoveta, which is a low trophic level species, is the largest single-species fishery in the world, and it exhibits very large fluctuations in catches (see Figure 1). This extreme variation masks the comparatively more subtle patterns in trophic level changes by the other fisheries of the world. We excluded large tunas and billfishes because much of their catch is taken in pelagic waters outside of the currently defined LMEs. Thus, including these landings from only part of their stock-exploitation ranges would

artificially inflate patterns of trophic levels, especially for recent decade, when tuna fisheries expanded substantially (Pauly and Palomares 2005). The trend in MTI for all LMEs combined (Figure 5, top) indicates a decline from a peak in the 1950s to a low in the mid 1980s. This can be attributed to ‘fishing down marine food webs’ (Pauly *et al.* 1998a, Pauly and Watson 2005), partly masked by an offshore expansion of the fisheries as indicated by the increasing FiB index (Figure 5, bottom). In the mid 1980s, the continued offshore expansion, combined with declining inshore catches has resulted in a trend reversal in the MTI, i.e. to the fishing down effect being completely occulted. Analyses at smaller scales (e.g., at the level of individual LMEs, see Sherman and Hempel 2008) confirm this.

Figure 5. Two indicators based on the trophic levels (TL) of exploited fish, here used to characterize the fisheries in all LMEs of the world combined. Top: Marine Trophic Index (MTI), being the trend of mean TLs; and bottom: corresponding trend of the Fishing-in-Balance (FiB) index, which is defined such that its increase in the face of stagnating or increasing MTI suggests a geographic expansion of the fisheries.



Considering the individual LMEs presented as examples, the Mean Trophic Index of the reported landings for the Newfoundland-Labrador Shelf LME remained high until the 1990s, when the cod stock began to collapse (Figure 6, top), a clear case of rapid ‘fishing down’ the food web in the LME (Pauly *et al.* 1998a). The FiB index

shows a similar trend (Figure 6, bottom), indicating that the reported landings did not compensate for the decline in the MTI over that period. However, these landings do not account for the discarded bycatch from the shrimp fishery, which now accounts for half of the value of the landings (Sherman and Hempel 2008).

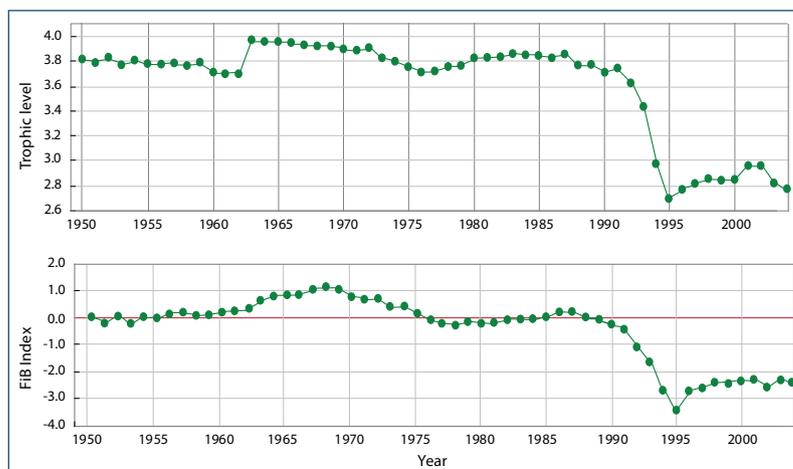


Figure 6. Mean trophic level (i.e. Marine Trophic Index) (top) and Fishing-in-Balance Index (bottom) in the Newfoundland-Labrador Shelf LME.

The Mean Trophic Index of the reported landings for the North Sea LME has shown a steady decline since 1970 (Figure 7, top), an indication of a steady ‘fishing down’ of the food web in the LME (Pauly *et al.* 1998a). The FiB index has been on a similar decline over the past three decades (Figure 7, bottom). Both indices thus correspond

with the detailed analysis by Froese and Pauly (2003), which was based on catch data starting in 1903.

The Mean Trophic Index of the reported landings for the Baltic Sea LME shows a distinct decline from the mid 1980s to 2004 (Figure 8, top), driven by the increased sprat landings.

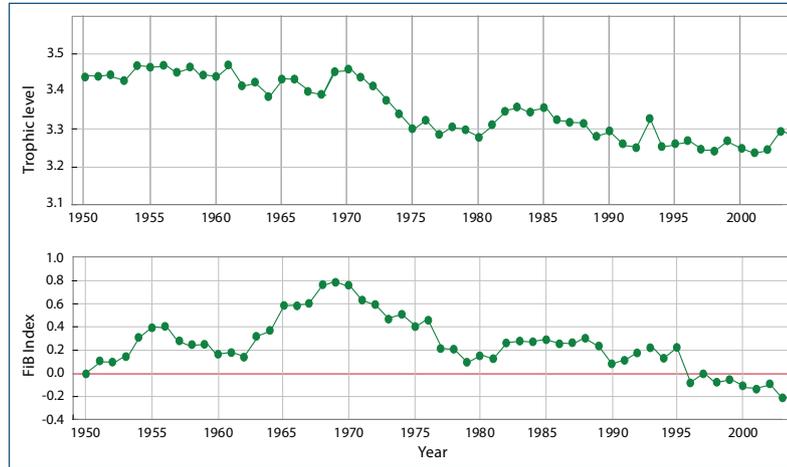


Figure 7. Mean Trophic Index (top) and Fishing-in-Balance Index (bottom) in the North Sea LME.

However, the simultaneous decline in Atlantic cod landings combines to create a case of ‘fishing down’ of the local food webs (Pauly *et al.* 1998a).

The rapid decline in the FiB index also supports this interpretation (Figure 8, bottom).

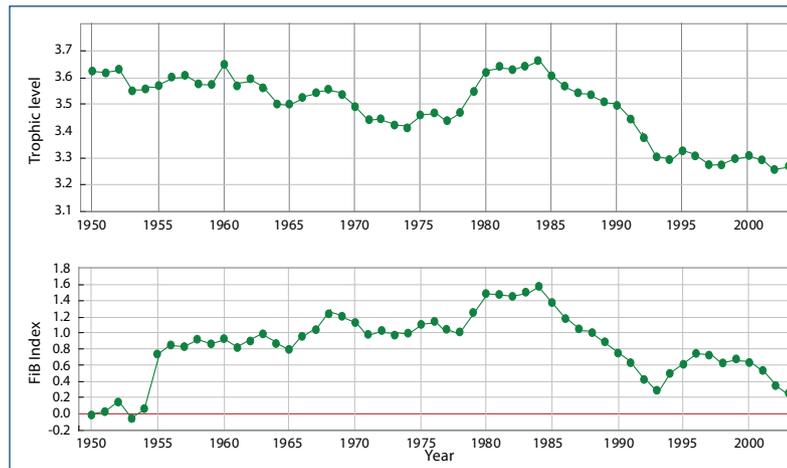


Figure 8: Mean Trophic Index (top) and Fishing-in-Balance Index (bottom) in the Baltic Sea LME.

Stock-Catch Status Plots

This graphical approach has its origin in the work of Grainger and Garcia (1996), who fitted time series of landings of the most important species in the FAO landings database with high-order polynomials, and evaluated from their slopes whether the fisheries were in their ‘developing’, ‘fully utilized’ or ‘senescent’ phases. Froese and Kesner-Reyes (2002) simplified these graphs by defining for any time series (taken as representing ‘stocks⁴’), five phases relative to the maximum reported landing in that time series:

1. **Undeveloped:** Year of landing is before the year of maximum landing, and landing is less than 10 percent of maximum landing;
2. **Developing:** Year of landing is before the year of maximum landing, and landing is between 10 and 50 percent of maximum landing;
3. **Fully exploited:** Landing is greater than 50 percent of maximum year’s landing;
4. **Overexploited:** Year of landing is after year of maximum landing, and landing is between 10 and 50 percent of maximum landing; and
5. **Collapsed:** Year of landing is after the year of maximum landing, and landing is below 10 percent of maximum landing.

Fisheries in a given area (e.g. LMEs) can therefore be diagnosed by plotting time series of the fraction of ‘stocks’ in any of these categories (Froese and Kesner-Reyes 2002). Such ‘stock number by status plots’ were used to document the state of the North Sea LME by Froese and Pauly (2004) and, more recently have been ap-

plied to all LMEs of the world (Pauly *et al.* 2008, Sherman and Hempel 2008).

Pauly *et al.* (2008) also proposed a variant of the above ‘stock number by status plots’ in the form of a ‘stock catch by status plot’, defined such that it presents the fraction of the reported landings ‘biomass’ (i.e. catch) that is derived from stocks in various phases of development (as opposed to the number of such stocks). Such a plot of relative ‘catch’ by status can present quite a different picture from the stock number by status plots. The combination of these two plots is now termed ‘*Stock-Catch Status Plots*’ (Pauly *et al.* 2008).

Figure 9 illustrates the dual nature of the newly derived *Stock-Catch Status Plots*, for the 53 LMEs combined. Overall, it suggests that 70 percent of global stocks within LMEs can be considered overexploited or collapsed (Figure 9, top). Nevertheless, these overexploited and collapsed stocks still provide 50 percent of the globally reported landings biomass, with the rest contributed by fully exploited stocks (Figure 9, bottom). This confirms the observation (e.g. Worm *et al.* 2006) that fisheries tend to affect biodiversity even more strongly than they affect biomass.

Turning to individual LMEs, the *Stock-Catch Status Plots* for the Newfoundland-Labrador Shelf LME show that over 60 percent of commercially exploited stocks in the LME are deemed collapsed with an additional 20 percent overexploited (Figure 10, top). Over 50 percent of the reported landings biomass is now supplied by fully exploited stocks (Figure 10, bottom).

4. Here, a ‘stock’ is defined as a time series of one species, genus or family for which the first and last reported landings are at least ten years apart, for which there are at least five years of consecutive catches and for which the catch per spatial entity (e.g. LME) is at least 1,000 tons (Pauly *et al.* 2008).

Figure 9. Paired *Stock-Catch-Status Plots* for all LMEs in the world combined, wherein the status of stocks, i.e., species with a time series of landings in an LME, is assessed using the following criteria (all referring to the maximum catch in the species time series): Developing (catches < 50 %); Fully exploited (catches > = 50 %); Overexploited (catches between 50 % and 10 %); Collapsed (catches < 10 %).

Top: Percentage of stocks of a given status, by year, showing a rapid increase of the number of overexploited and collapsed stocks.

Bottom: Percentage of catches extracted from stocks of a given status, by year, showing a slower increase of the percentage of catches that originate from overexploited and collapsed stocks. Note that 'stocks', i.e. individual landings time series, only include taxonomic entities at species, genus or family level, i.e. higher and pooled groups have been excluded.

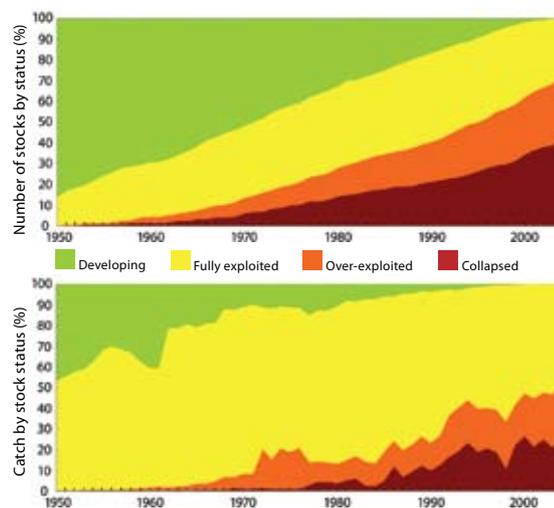
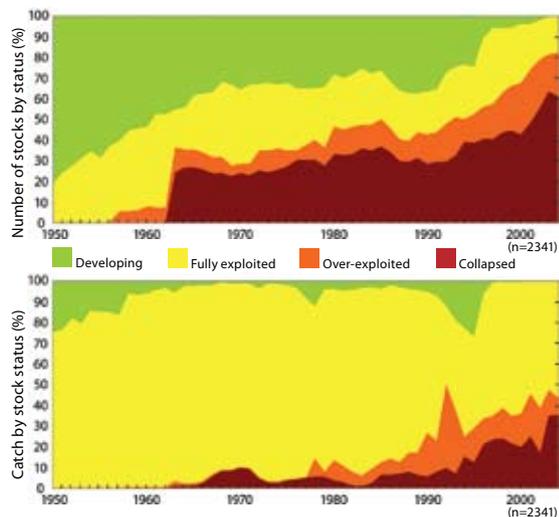


Figure 10. *Stock-Catch Status Plots* for the Newfoundland-Labrador Shelf LME, showing the proportion of developing (green), fully exploited (yellow), overexploited (orange) and collapsed (purple) fisheries by number of stocks (top) and by catch biomass (bottom) from 1950 to 2004.

Note that (n), the number of 'stocks', i.e. individual landings time series, only include taxonomic entities at species, genus or family level, i.e. higher and pooled groups have been excluded (see Pauly *et al.* in press for definitions).



The *Stock-Catch Status Plots* for the North Sea LME, based on the first analysis of an LME using such plots (Froese and Pauly 2003), indicate that the number of collapsed and overexploited stocks have been increasing, accounting for close to 80 percent of all commercially exploited stocks in the North Sea (Figure 11, top). A majority of the reported landings biomass, particularly in

recent years, is supplied by overexploited stocks (Figure 11, bottom).

The *Stock-Catch Status Plots* for the Baltic Sea LME indicate that over 60 percent of the fished stocks in the LME have collapsed (Figure 12, top), but that the majority of the catch is supplied by fully exploited stocks (Figure 12, bottom), likely due to the large European sprat catches.

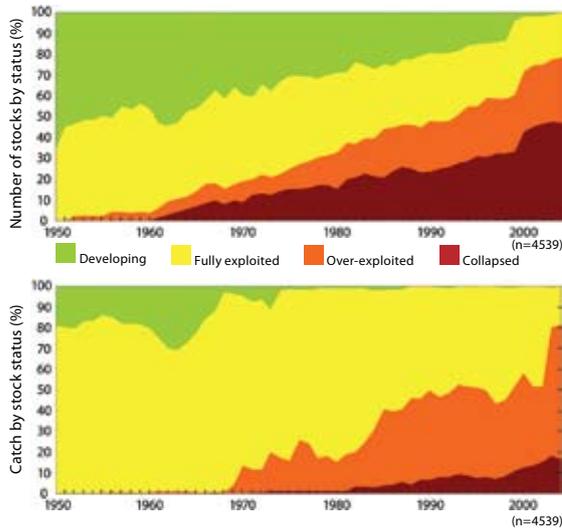


Figure 11: *Stock-Catch Status Plots* for the North Sea LME, showing the proportion of developing (green), fully exploited (yellow), overexploited (orange) and collapsed (purple) fisheries by number of stocks (top) and by catch biomass (bottom) from 1950 to 2004.

Note that (n), the number of 'stocks', i.e. individual landings time series, only include taxonomic entities at species, genus or family level, i.e. higher and pooled groups have been excluded (see Pauly *et al.* in press for definitions).

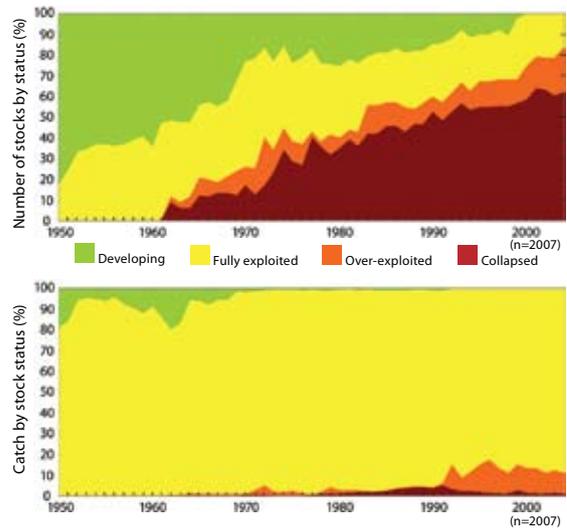


Figure 12: *Stock-Catch Status Plots* for the Baltic Sea LME, showing the proportion of developing (green), fully exploited (yellow), overexploited (orange) and collapsed (purple) fisheries by number of stocks (top) and by catch biomass (bottom) from 1950 to 2004.

Note that (n), the number of 'stocks', i.e. individual landings time series, only include taxonomic entities at species, genus or family level, i.e. higher and pooled groups have been excluded (see Pauly *et al.* in press for definitions).

Conclusions

Indices such as those presented here, or others, usually cannot be used to manage fisheries (or fleets) on a year to year (or tactical) basis. They can be used, however, to indicate broad, ecosystem- or indeed planet-wide trends, similar to e.g. atmospheric carbon dioxide concentration used to monitor our success – or lack thereof – in combating global warming. They indicate trends that must be addressed by humanity as a whole if we are to continue extracting benefits from – in this case – marine ecosystems.

The fishing industry is, on its own, incapable of turning around the trends demonstrated above. In fact, many industry representatives and their supporting cast in academia deny or fail to recognize that there is a global problem (see e.g. Bigot 2002, Lomborg 2001). There are other social forces, however, which can and will play an increasing role in the international debates on fisheries, foremost among those the community of Non-Governmental Organizations devoted to maintaining or re-establishing ‘healthy’ marine ecosystems, and to push toward ecosystem-based fisheries management (Pikitch *et al.* 2004), irrespective of how these terms may end up being implemented in practice. Hence the public debate about fisheries, unheard of two decades ago, and hence the need for conservation biologists to debunk those who deny the need for action. One of these is the above-cited Lomborg, who in his *Sceptical Environmentalist*, inferred from increasing global catch figures reported by the FAO that the underlying ecosystems must be in good shape. However, we now know that the apparent

increases of global fisheries catches in the 1990s were due to China massively over-reporting its catches to the FAO, and global catches are in fact declining (Watson and Pauly 2001, Pauly 2002). And we also know that catches can remain high (and in fact usually do) when stocks collapse, as illustrated by the northern cod off Eastern Canada, which yielded good catches until the fishery had to be closed because there were literally no fish left (Myers *et al.* 1997).

Lomborg (2001), here representing a large number of ill-informed commentators, also suggested that ‘aquaculture’ could help compensate for overfishing. He believes “it appears of minor importance whether the consumer’s salmon stems from the Atlantic Ocean or a fish farm”. The problem here is that ‘aquaculture’ covers two fundamentally different kinds of operations; let’s call them Aquaculture A and B. Aquaculture A, devoted to the farming of bivalves such as oysters or mussels, or to freshwater fish such as carp or tilapia, relies on plant matter (phytoplankton in the sea or in ponds, sometimes supplemented by agricultural by-products in the case of freshwater fishes), to generate a net addition to the fish food supply available to consumers. Moreover, because Aquaculture A is based predominantly in developing countries (mainly in China, but also in countries such as the Philippines, or Bangladesh), it supplies cheap animal protein right where it is needed (New 2002).

Aquaculture B is the farming of carnivorous fish such as salmon or seabass, and increasingly, the fattening of wild caught bluefin tuna, e.g. in the Mediterranean. In nature, salmon, seabass

and bluefin tuna have high trophic levels, ranging from 3.5 to 4.5 (see references in Table 1, and www.fisbbase.org). When fed only vegetable matter, e.g. soy meal, salmon do not grow well, and end up looking and tasting like tofu. As for tuna, there is no point even trying to feed them with anything but fish. What this implies is that the more aquaculture B is undertaken, the less cheap yet healthy (high omega-3 oils) fish such as sardine, herring, mackerel and anchovies there will be for humans to buy and eat. Aquaculture B does not reduce the pressure on wild fish stocks: it increases it (Naylor *et al.* 2000). It has led to massive imports, by developed countries, where Aquaculture B predominates, of fish-meal from fishes caught and ground up in developing countries, thus exacerbating fishing pressures in these countries. We will not elaborate on the coastal pollution and diseases emanating from the uneaten food and feces of these marine feedlot operations, which responsible practitioners see as a major constraint to the development of the industry (New 2002).

One reason why the practitioners of aquaculture B can get away with all this is that the public at large assumes their operations to be similar to those of Aquaculture A, and that they add to the global fish supply. The continued growth of Aquaculture B will be more difficult once the distinction becomes clear to the public.

There is still time for fisheries, but only if they are reinvented not as the source of an endlessly growing supply of fish for an endlessly growing human population, but as provider of a healthy complement to predominantly grain-based diets. Particularly, fisheries cannot remain a free-for-all for pillaging distant water fleets; they can however, become a regular source of income for communities whose members act in accord with the finite nature of marine resources (Pitcher 2001). Two key elements of such reinvented fisheries will be their considerably smaller size, and their reliance on fish biomass being exported from numerous, large marine reserves, the protected ocean areas that we must establish and enforce if we are to allow marine ecosystems and the species therein to rebuild some of their past abundance, and to share this with us.

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