FISHING DOWN MARINE FOOD WEB: IT IS FAR MORE PERVASIVE THAN WE THOUGHT

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ABSTRACT

The widespread call for a transition toward "ecosystem-based" fisheries manage-ment implies the development and testing of sustainability indicators suitable for inferences on the status of the ecosystems within which fisheries resource species are embedded. The mean trophic level (TL) of fisheries catches has been shown to allow for such inferences, leading in the process to the identification of global trends toward catches being increasingly dominated by low-TL species, a process now known as "fishing down marine food webs." However, for inferences from TL trends to be accurate, taxonomic and especially geographic over-aggregation of the underlying catch data must be avoided. Accounting for these strong sources of bias suggests that the fishing down effect is far more pervasive than previously thought, and in fact occurs in areas where initial analyses failed to detect it. This confirms the common verdict of absent sustainability for most fisheries of the world. This is il-lustrated here by five brief case studies, of which three also document a new method for estimating ecosystem transfer efficiency, under a set of specific conditions, from time series of catches and their corresponding mean TL. Some suggestions are given on how work in this area might proceed.

While pre-industrial fisheries had the capacity to extirpate some freshwater and coastal fish populations, as evidenced in the sub fossil and archeological records (Jackson et al., 2001; Pitcher, 2001), it is only since the advent of industrial fishing that the sequential depletion of coastal, then offshore populations of marine fish has become the standard operating procedure (Ludwig et al., 1993).

In the late 19th century, in the North Sea, where British steam trawlers were first deployed, it took only a few years for the accumulated coastal stocks of flatfish (and other groups) to be depleted, and for the trawlers to be forced to move on to the cen-tral North Sea, then farther, all the way to Iceland (Cushing, 1988).

Similar expansions occurred in other parts of the world, and this led, after World War II, to massive increases of fisheries catches in the North Atlantic and the North Pacific, as well as in southeast Asia (Pauly and Chua, 1988; Pauly, 1996). By the late 1990s, the last large shelf areas previously not subjected to trawling had been de-pleted. All that is left for the expansion of bottom trawling are the last few seamounts and a few deepwater demersal fish populations, whose extremely slow somatic growth rates, associated with great individual fish longevity, essentially preclude sustainable exploitation. Hence, in the absence of legal protection, they are subjected to "pulse-fishing" by distant water fleets of various industrial countries; i.e., to rapid depletion of their biomass, without even the pretense of some form of sustainability (Gordon, 1999).

Similarly worrying trends are occurring in open-water ecosystems, where longlin-ing for tuna and other large pelagic fishes depletes these systems of large predators, including sharks, now feeding an insatiable soup fin market. Also, purse-seining around floating objects (i.e., natural or artificial fish aggregation devices) has made previously inaccessible small tunas and associated organisms vulnerable to fishing

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(Floyd and Pauly, 1984), thus prompting fears of the drastic decline of fish populations previously thought largely immune to overfishing (Myers and Worm, 2003).

The change in demersal and pelagic ecosystem structure resulting from such serial depletions can be illustrated in various ways. One of these is through the mean trophic level (TL) of fish in fisheries catches. These are declining throughout the world, implying that, globally, fisheries increasingly rely on fish originating from the bottom of marine food webs; i.e., on the prey of larger fishes (Pauly et al., 1998a,b; NRC, 1999).

The reality of this process, now known as "fishing down marine food webs," initially demonstrated based on FAO's (unadjusted) global data set, has since been confirmed by detailed analyses of various fishing grounds for which longer and more detailed catch time series are available (e.g., Greek waters: Stergiou and Koulouris, 2000; Cuban waters: Baisre, 2000; Canadian waters: Pauly et al., 2001; the Celtic Sea: Pinnegar et al., 2002; Icelandic waters: Valtysson and Pauly, 2003).

These findings enabled an expansion on the brief response (Pauly et al., 1998b) to the concerns raised in a comment by Caddy et al. (1998), thereby demonstrating that: 1) the relative abundances of various species in the catch statistics used for these analyses correlates sufficiently with the relative abundances of these species in the underlying ecosystems to justify the use of catch statistics for ecosystem inferences (see, e.g., Pinnegar et al., 2002); 2) taxonomic over-aggregation, as occurs in the FAO catch dataset tends to mask the fishing down process, which was thus underestimated in the original analysis (see Pauly and Palomares, 2000, 2001, and data presented below); 3) explicit consideration of within-species TL changes caused by change in size composition of exploited populations leads to trends of TL declining more strongly than when ontogenic diet changes are ignored (Pauly and Palomares, 2000; Pauly et al., 2001; Valtysson and Pauly, 2003); and that bottom-up effects (e.g., localized increases in primary productivity) can be detected when they occur (via the *FiB* index; i.e., Fishing in Balance, of Pauly et al., 2000), and thus can be considered when analyzing time series in term of fishing down effects.

Perhaps more importantly, these analyses allowed the identification of another, strong source of bias, so far not considered when quantifying "fishing down;" i.e., the masking effect resulting from unwittingly combining data from incompatible areas, notably catches from shelf and oceanic fisheries. This effect is particularly strong in areas where tuna fisheries have developed immediately adjacent to demersal fisheries on narrow shelves, leading to mean TL estimates that are determined not by ecological changes within either of these fisheries, but by the changing ratios of the magnitudes of their respective catches.

Based on the global FAO dataset, this contribution assumes the existence of the fishing down effect and aims to evaluate its strength once the issue of geographic over-aggregation and related problems are accounted for. The goal here is to provide the basis for a reliable estimate of the mean rate at which TL are declining globally, for use as reference point in local studies. Also, we use the opportunity provided by the presentation of series of mean TL and related indicators to describe a new approach for estimating the efficiency of biomass transfer from the lower to higher trophic levels of an exploited ecosystem.

Material and Methods

Trophic levels express the position of organisms within the food webs that largely define aquatic ecosystems. Their values are set as one in plants and detritus, two in herbivores and detritivores (first-level consumers), three in second-level consumers, etc. However, real consumers, which tend to have catholic diets, do not usually have TL with integer values (Odum and Heald, 1975). The definition of TL for any consumer species i is thus:

$$TL_i = 1 + \sum_j \left(TL_j \cdot DC_{ij} \right)$$

Eq. 1

where TL_j is the "fractional" (i.e., non-integer) trophic level of the prey j, and DC_{ij} represents the fraction of j in the diet of i.

Thus defined, the TL of most fish and other aquatic consumers can take any value between 2.0 and 5.0, the latter being rare even in large fish (Cortés, 1999), occurring only in specialized predators of marine mammals, such as killer whales or polar bears (Pauly et al., 1998c). The values of TL_i for fish species and other taxa in the FAO statistics used for the analyses below are documented in the "ISSCAAP Table" of FishBase (Froese and Pauly, 2000). Also note that Equation (1) defines TL as measurable entities rather than "concepts" (Rigler, 1975); as such, they can be cross-validated using different methods, notably mass-balance models of trophic fluxes in ecosystems (Christensen and Pauly, 1993; Pauly et al., 2000), and the ratios of ^{15}N to ^{14}N in consumers' tissues (Kline and Pauly, 1998).

Mean trophic level from an area (or ecosystem) are computed, for any year y from:

$$TL_y = \sum_i \left(TL_i \cdot Y_{iy} \right) \Big/ \sum_i Y_{iy}$$

Eq. 2

where Y_{iy} is the catch of species (group) i in year y, and TL_i is defined as in Equation (1).

Given that biological production is higher at lower than at higher TL, fisheries catches, initially at least, will tend to increase when TL decline (i.e., when the fisheries target species lower in the food web; Pauly et al., 1998a). This led Pauly et al. (2000) to suggest a *FiB* index which, given an estimate of the biomass (or energy) transfer efficiency (TE) between TL, maintains a value of zero when a decrease in TL is matched by an appropriate catch increase (and conversely when TL increases), and deviates from zero otherwise. The *FiB* index is defined, for any year y, by

$$FiB_y = \log \left\{ \left[Y_y \cdot \left(1/TE \right)^{TL_y} \right] \Big/ \left[Y_0 \cdot \left(1/TE \right)^{TL_0} \right] \right\}$$

Eq. 3

where Y_y is the catch at year y; TL_y is the mean trophic level of the catch at year y; Y_0 is the catch and TL_0 the mean trophic level of the catch at the start of the series being analyzed. Note that the *FiB* index is designed such that it does not vary during those periods when changes in TL are matched by catch changes in the opposite direction; i.e., periods within a time series where the variation in the *FiB* index appears to follow a horizontal progression. Thus, an increase of the *FiB* index indicates that the underlying fishery is expanding beyond its traditional fishing area (or ecosystem). Conversely, a decrease indicates a geographic contraction, or a collapse of the underlying food web, leading to the "backward-bending" plots of TL vs catch originally presented in Pauly et al. (1998a).

In marine ecosystem studies, TE, which usually ranges from 0.025−0.25, is often set at 0.1, a mean value recently re-validated through the analysis of a large number of quantitative food webs (Christensen and Pauly, 1993; Pauly and Christensen, 1995).

However, we can do better than using a mean, if we assume, with a number of authors (review in Pauly, 1996), that fisheries, within a given ecosystem, take a similar fraction of bio-

logical production over the range of trophic levels from which fisheries catches are extracted (from TL = 2.0 to about 4.5 if "fisheries" for algae are not considered). In this case, an ecosystem-specific mean estimate of TE can be obtained from any run of catch and mean TL values within which *FiB* remains equal or close to zero; i.e., from the slope (b) of the equation:

$$TL = a + b \cdot \log\left(Y_y\right)$$

<div align="right">Eq. 4</div>

where $TE = 10^{1/b}$. This follows the logic of equation (3), which implies that the rate at which catches increase when TL decrease (or vice versa) is a function of TE.

Results and Discussion

Taxonomic Over-Aggregation.—Figure 1 (top) documents the level of taxonomic resolution in FAO statistics for two regions representing the range of resolution occurring in the 18 existing FAO areas. Thus, while well over 80% of the catches from the northwest Atlantic are reported at species level and over 95% at genus level, only 10% of the catches from the western-central Atlantic are reported at species level and slightly over 30% at genus level. Because of the large contribution to the global catch of tropical areas with low taxonomic resolution, we estimated that globally, <60% of FAO's global catch statistics are reported at species and only 70% at genus level (central line, top of Fig. 1).

When a large fraction of the catch is reported at family or higher taxonomic level, the downward trend in TL is underestimated (see e.g., fig. 2 in Pauly and Palomares, 2001). This is here illustrated in Figure 1 (bottom) where the rate of decline of mean TL against time is plotted for all FAO areas against the percentage of catch reported at species level. Thus, due to taxonomic over-aggregation in the underlying dataset, the first approximation (in Pauly et al., 1998a) of the global rate of TL decline underestimated the strength of the "fishing down effect."

This is further illustrated in Figure 2 (top), which shows a surprisingly regular decline in the mean TL of tunas and billfishes (ISSCAAP group 36 in the FAO database; see Table 1) caught in the oceans of the entire world. This phenomenon, though not very strong in absolute term, implies that tuna and billfish species with lower TL now contribute relatively more to catches than they did before. This trend was not detected earlier because the fish from that category were previously part of the catch data sets used for trend analyses by FAO areas, and not analyzed separately.

As noted by Caddy et al. (1998), trophic levels vary between different populations of a species and during the ontogeny of individuals. Our use of the median trophic levels for populations with more than one estimate in FishBase 2000 (see Froese and Pauly, 2000) was to ensure that our analyses be based on conservative estimates (see Table 1). We did not conduct sensitivity analyses of between population variability of trophic levels but we believe that the effect would have been inconsequential given the strong decline in the catch-weighted average size of ISSCAAP group 36 from 1950–1998 (based on the length also given in Table 1; data not shown). As for ontogenic effects, they not only tend to be very small, but also tend to increase the fishing down effect, since most fish such as those included in ISSCAAP group 36 tend to reach higher TL as they get older (Pauly et al., 2001).

Also note that the decline of mean TL in Figure 2 (top) has been more than compensated for by the increase in catches, as illustrated by the increasing values of the
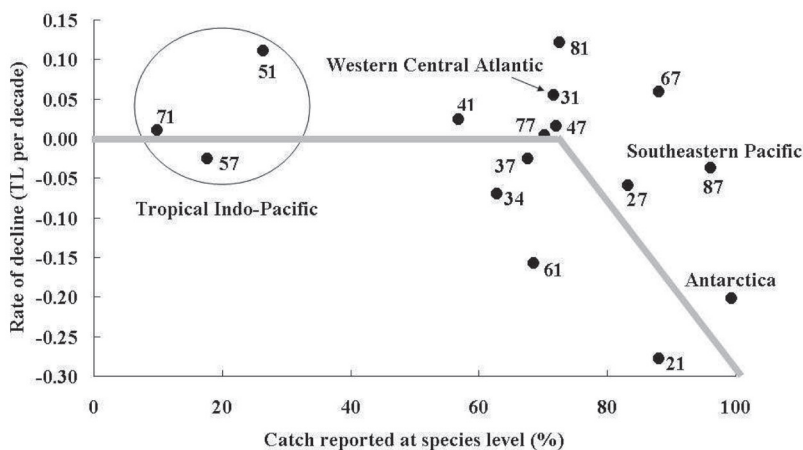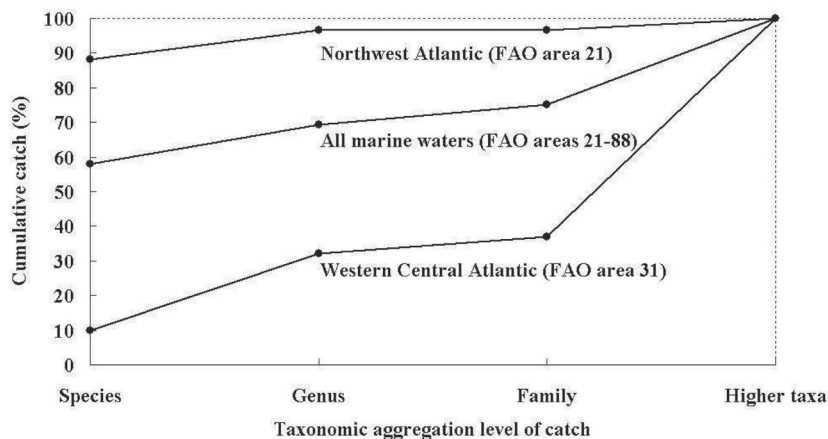
Figure 1. Taxonomic aggregation levels in the FAO marine catch database, and the effect on the estimation of TL trends; based on FAO data for 1997. Top: two examples representative of well disaggregated (area 21) and over-aggregated data (area 31), both jointly bracketing the world average. Bottom: slope of plots of mean TL vs time for all of the 18 FAO statistical areas, illustrating that taxonomic over-aggregation (low values along ordinate axis) tends to be associated with zero or higher slopes (suggesting an absence of the fishing down effect). Note that the point for Antarctica represents FAO areas 48, 58, and 88.

FiB index (Fig. 2, bottom), suggestive of a steady geographic expansion of the fisheries over the entire period considered here, as was indeed the case (Fonteneau, 1998; Myers and Worm, 2003).

Geographic Over-Aggregation.—It could be argued that including tuna and billfishes in analyses involving a wide range of neritic fishes, over-aggregates species in a taxonomic sense, rather than in a bio-geographic sense. A better example of geographic over-aggregation thus is the case of the western-central Atlantic (Fig. 3, top), whose mean TL has apparently been increasing over recent time (see also Fig. 1, bottom).

Decomposing this area into two components with distinct fisheries and history of exploitation, one representing the U.S. (Atlantic coast south of Chesapeake Bay and northern Gulf of Mexico), the other the Greater Caribbean area, leads to two trend
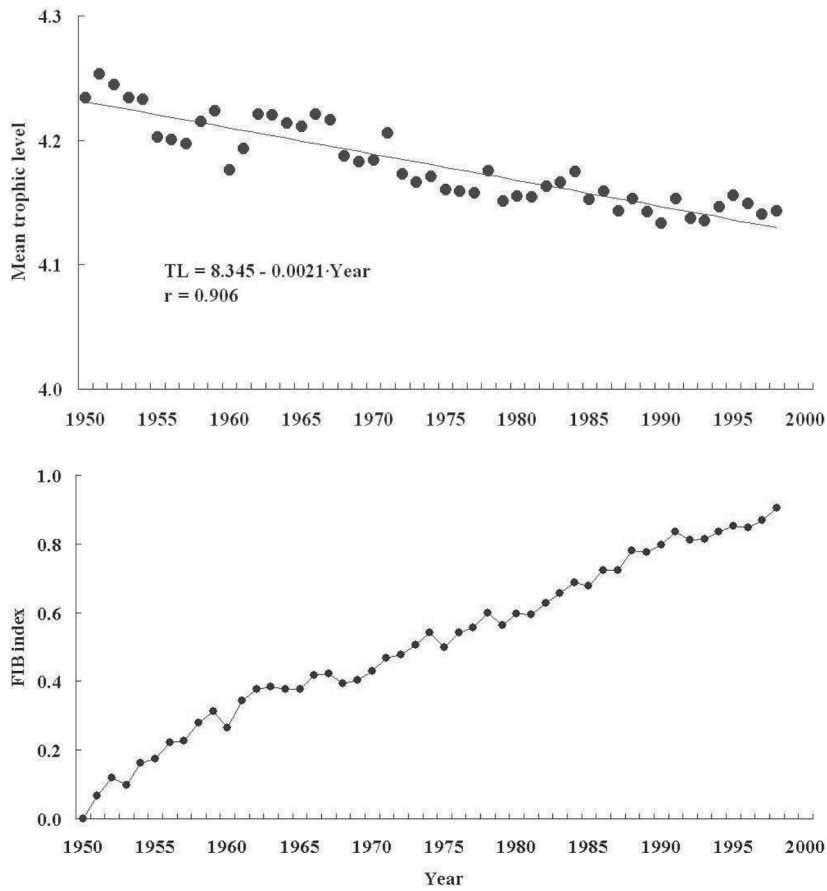
Figure 2. Analysis of FAO global statistics for tuna and billfishes (ISSCAAP group 36; Table 1). Top: trend of mean TL, indicative of a transition toward a greater contribution of smaller tuna and billfish species in the total catch. Bottom: *FiB* index for the same data (computed with TE = 10%), whose steady increase implies a geographic expansion of the fisheries (see text).

lines, each documenting a clear "fishing down" effect (Fig. 3, bottom). Here, a relative increase in the catches (which serve as a weighting factor when computing mean TL) from the southern part of FAO area 31 completely masked the underlying downward trend(s) in TL. This leads to the conclusion that analyses of mean TL should combine data from a large area only if their fisheries remain similar over time in terms of species composition and catch levels.

Regional Analyses and the Estimation of Transfer Efficiencies.—Figure 4 (top) documents the trends in catches and mean TL in FAO area 21 (northwest Atlantic), a region initially dominated by small pelagic fisheries catching low-TL fishes. This was followed by a transition toward high-TL piscivores; i.e., TL ≥ 3.75 as used in Christensen et al. (2003), notably cod (*Gadus morhua* Linnaeus, 1758) in increasingly deeper waters (i.e., a geographic expansion), whose stocks then collapsed, leading to a transition towards low-TL invertebrates.

The corresponding *FiB* index series, in Figure 4 (center), shows the geographic expansion alluded to above in form of a run of values positive indices, starting from the

Table 1. Median trophic level and maximum length (from FishBase 2000; see Froese and Pauly 2000) of species (groups) used in trend analysis of ISSCAAP group 36 ("tuna and billfishes", including tuna-like fishes and large mackerels; see Fig. 2).

| Scientific name(s)	Common name(s)	Aggreg. level	Trophic level	SL$_{max}$ (cm)
Auxis spp.	Frigate and bullet tunas	Genus	4.00	164
Istiophoridae	Marlins, sailfishes, pearfishes	Family	4.20	263
Scomberomorus spp.	Seerfishes nei	Genus	3.90	128
Scombroidei	Tuna-like fishes nei	Families	3.80	162
Thunnini	Tunas nei	Families	3.70	115
Thunnus spp.	Tuna	Genus	3.70	303
Xiphiidae	Swordfishes	Family	4.20	369
Acanthocybium solandri (Cuvier, 1832)	Wahoo	Species	4.50	195
Allothunnus fallai Serventy, 1948	Slender tuna	Species	4.10	98
Auxis rochei (Risso, 1810)	Bullet tuna	Species	4.13	164
Auxis thazard (Lacepède, 1800)	Frigate tuna	Species	4.34	54
Cybiosarda elegans (Whitley, 1935)	Leaping bonito	Species	4.00	37
Euthynnus affinis (Cantor, 1849)	Kawakawa	Species	4.47	93
Euthynnus alletteratus (Rafinesque, 1810)	Little tunny	Species	3.80	100
Euthynnus lineatus Kishinouye, 1920	Black skipjack	Species	4.00	78
Gasterochisma melampus Richardson, 1845	Butterfly kingfish	Species	4.00	153
Grammatorcynus bicarinatus (Quoy & Gaimard, 1825)	Shark mackerel	Species	4.00	102
Grammatorcynus bilineatus (Rüppell, 1836)	Double-lined mackerel	Species	4.00	93
Gymnosarda unicolor (Rüppell, 1836)	Dogtooth tuna	Species	4.00	180
Istiophorus albicans (Latreille, 1804)	Atlantic sailfish	Species	4.20	258
Istiophorus platypterus (Shaw, 1792)	Indo-Pacific sailfish	Species	4.20	279
Katsuwonus pelamis (Linnaeus, 1758)	Skipjack tuna	Species	4.37	100
Makaira indica (Cuvier, 1832)	Black marlin	Species	4.20	367
Makaira mazara (Jordan & Snyder, 1901)	Indo-Pacific blue marlin	Species	4.20	367
Makaira nigricans Lacepède, 1802	Atlantic blue marlin	Species	4.20	346
Orcynopsis unicolor (Geoffroy Saint-Hilaire, 1817)	Plain bonito	Species	4.00	121
Sarda australis (Macleay, 1881)	Australian bonito	Species	4.00	93
Sarda chiliensis (Cuvier, 1832)	Eastern Pacific bonito	Species	4.50	84
Sarda orientalis (Temminck & Schlegel, 1844)	Striped bonito	Species	4.00	95
Sarda sarda (Bloch, 1793)	Atlantic bonito	Species	3.80	85
Scomberomorus brasiliensis Collette, Russo & Zavala-Camin, 1978	Serra Spanish mackerel	Species	4.00	116
Scomberomorus cavalla (Cuvier, 1829)	King mackerel	Species	4.50	151
Scomberomorus commerson (Lacepède, 1800)	Narrow-barred Spanish mack	Species	4.50	205
Scomberomorus concolor (Lockington, 1879)	Monterey Spanish mackerel	Species	4.00	71
Scomberomorus guttatus (Bloch & Schneider, 1801)	Indo-Pacific king mackerel	Species	4.00	71
Scomberomorus koreanus (Kishinouye, 1915)	Korean seerfish	Species	4.00	140
Scomberomorus lineolatus (Cuvier, 1829)	Streaked seerfish	Species	4.00	74
Scomberomorus maculatus (Mitchill, 1815)	Atlantic Spanish mackerel	Species	4.50	77
Scomberomorus multiradiatus Munro, 1964	Papuan seerfish	Species	4.00	33
Scomberomorus munroi Collette & Russo, 1980	Australian spotted mackerel	Species	4.00	93
Scomberomorus niphonius (Cuvier, 1832)	Japanese Spanish mackerel	Species	4.00	93
Scomberomorus plurilineatus Fourmanoir, 1966	Kanadi kingfish	Species	4.00	112
Scomberomorus queenslandicus Munro, 1943	Queensland school mackerel	Species	4.50	93
Scomberomorus regalis (Bloch, 1793)	Cero	Species	4.48	150
Scomberomorus semifasciatus (Macleay, 1883)	Broad-barred king mackerel	Species	4.00	112
Scomberomorus sierra Jordan & Starks, 1895	Pacific sierra	Species	4.00	90
Scomberomorus sinensis (Lacepède, 1800)	Chinese seerfish	Species	4.00	186
Scomberomorus tritor (Cuvier, 1832)	West African Spanish mackerel	Species	4.00	82
Tetrapturus albidus Poey, 1860	Atlantic white marlin	Species	4.20	246
Tetrapturus angustirostris Tanaka, 1915	Shortbill spearfish	Species	4.20	189
Tetrapturus audax (Philippi, 1887)	Striped marlin	Species	4.58	287
Tetrapturus belone Rafinesque, 1810	Mediterranean spearfish	Species	4.20	197
Tetrapturus georgei Lowe, 1841	Roundscale spearfish	Species	4.20	151
Tetrapturus pfluegeri Robins & de Sylva, 1963	Longbill spearfish	Species	4.20	205
Thunnus alalunga (Bonnaterre, 1788)	Albacore	Species	4.46	121
Thunnus albacares (Bonnaterre, 1788)	Yellowfin tuna	Species	3.70	259
Thunnus atlanticus (Lesson, 1831)	Blackfin tuna	Species	4.13	93
Thunnus maccoyii (Castelnau, 1872)	Southern bluefin tuna	Species	4.00	228
Thunnus obesus (Lowe, 1839)	Bigeye tuna	Species	4.49	219
Thunnus thynnus (Linnaeus, 1758)	Northern bluefin tuna	Species	4.43	376
Thunnus tonggol (Bleeker, 1851)	Longtail tuna	Species	4.06	126
Xiphias gladius Linnaeus, 1758	Swordfish	Species	4.55	369
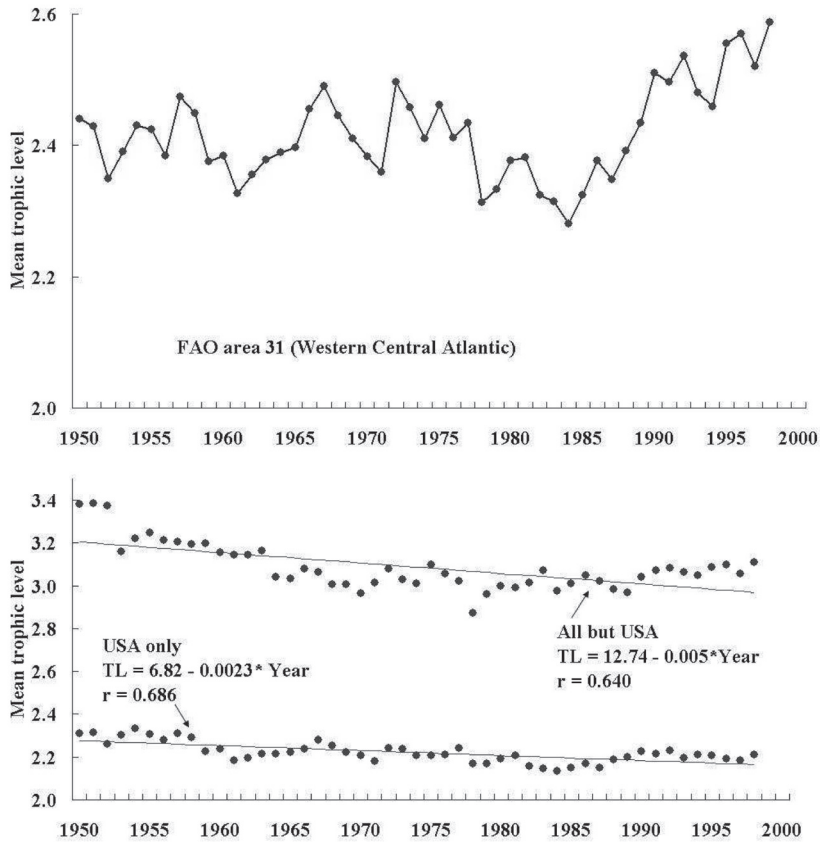
Figure 3. Analysis of FAO marine fisheries catch statistics for area 31 (western central Atlantic. Top: analysis based on aggregated data for the entire area; note absence of any "fishing down" signal (see also Fig. 1). Bottom: separation of the data for area 31 into two subset leads to identification of two clear "fishing down" trends, previously masked by geographic over-aggregation). Both regressions have slopes significantly different from zero at P = < 0.0001.

mid 1950s and lasting to the mid 1970s. The following period (1977–1990) is characterized by a near constant *FiB* index, wherein any change in mean TL is matched by a change in catches in opposite direction. This period, which preceded the collapse of large predatory stocks (especially cod) and their supporting food webs in the 1990s, can thus be used for estimating TE for the upper trophic levels of the ecosystem, according to the methodology outlined above; i.e., equation (4). The resulting plot is shown in Figure 4 (bottom), and it leads to an estimate of TE = 6.8%.

The catches in FAO area 87 (Chile and Peru) have long been dominated by the Peruvian anchoveta (*Engraulis ringens* Jenyns, 1842) which has a very low TL level (2.2, see Froese and Pauly, 2000). Hence, mean TL trends based on all catches from area 87 are largely determined by the fluctuation of the anchoveta catches (Fig. 5, top, open circles), themselves driven by a succession of El Niño events (Muck, 1987). "Fishing down" does appear, however, when the trend for all species except anchoveta is plotted (Fig. 5, top, solid circles), with the slight TL increase occurring in the 1990s being due to massive catches of high-TL horse mackerel, *Trachurus pictura-*
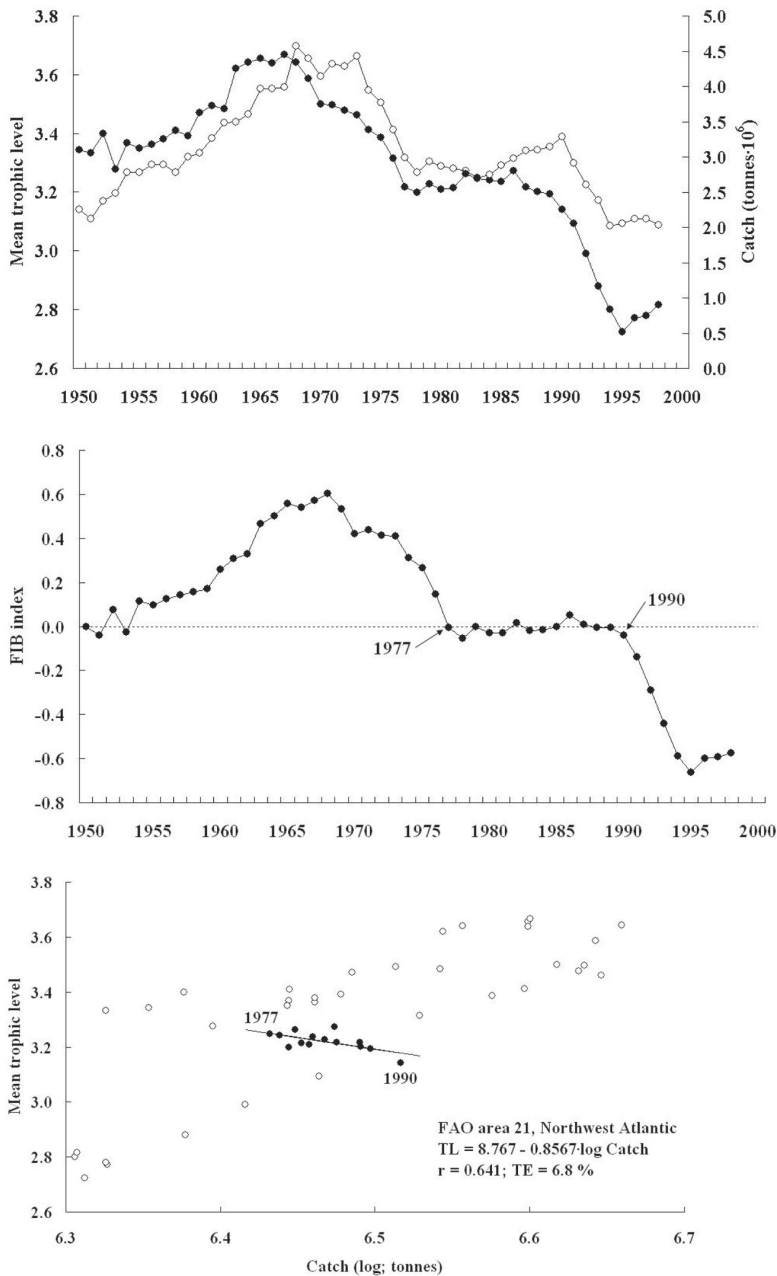
Figure 4. Analysis of the FAO marine fisheries catch statistics for area 21 (western north Atlantic. Top: Catch (open circles) and mean TL (solid circles) roughly parallel each other, and, in this form, do not lead to any new inference. Center: Re-expressed as *FiB* index (with TE = 10%), the same data show that the catch increases from the mid 1950s to mid 1970s were probably due to a geographic expansion. Moreover, the period from 1977 to the collapses of the 1990s exhibits constants values of the *FiB* index, suggesting that the fisheries were "in balance" during that period (Turning Points and Pairwise Rank tests of randomness (see Kanji, 1993) were used to establish absence of FiB trends in that period; see text). Bottom: the "flat" period identified above selection of the data points used for estimating TE = 6.8%.
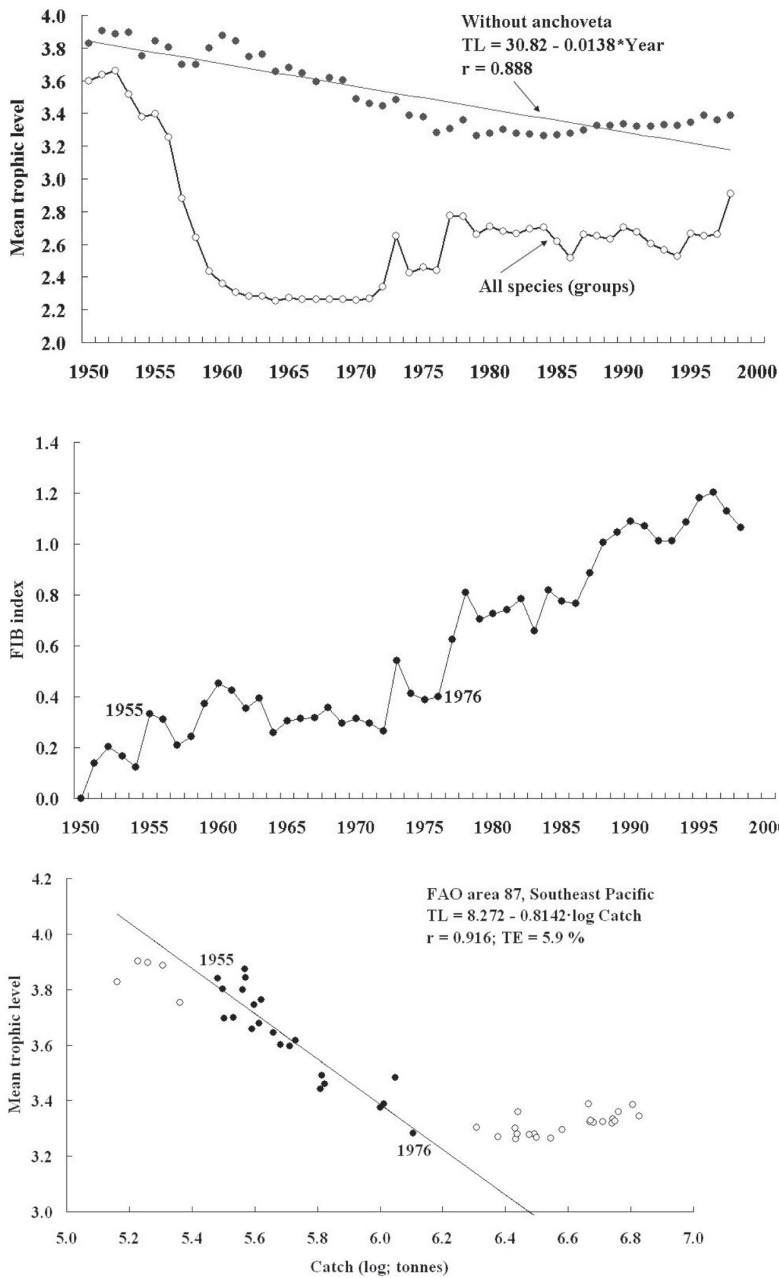
Figure 5. Analysis of the FAO marine fisheries catch statistics for area 87 (southeastern Pacific). Top: The mean TL trend computed with anchoveta (open circles) is uninformative, due to it being completely dominated by that species. A clear "fishing down" signal emerges, once that species is removed (the trend deviation in the 1990s is due to horse mackerel, a high-TL oceanic species). Center: the *FiB* index for the catch data excluding anchoveta show a steady increase, due to increasing horse mackerel catches from oceanic waters. The period from 1955–1976, prior to this offshore expansion, does not exhibit significant change, however (same test as for Fig. 4). Bottom: Using the data from 1955–1976 leads to an estimate of TE = 5.9% for the transfer efficiency between the higher trophic levels of the Humboldt Current system.

tus (Bowdich, 1825) an oceanic species (Parrish, 1987). The *FiB* index for the period 1955–1976 is relatively flat, and hence the catch and mean TL series for this period were used to estimate a transfer efficiency of 5.9% (Fig. 5, bottom).

Given the encouraging results from the analyses of *FiB* trends in FAO areas 21 and 87, we conclude our case studies with an analysis of the FAO data for China in FAO area 61 (Pacific Northwest), based on the catch data submitted by China to FAO. Watson and Pauly (2001) have suggested that the marine fisheries catches reported by China (Fig. 6, top, open circles) are most probably biased upward, especially from the mid 1990s on, even if account is taken of the Chinese distant-water fisheries (Pang and Pauly, 2001). Moreover, these data are taxonomically over-aggregated, with about 60% of the reported catch being categorized as "miscellaneous" fishes, mollusks, or crustaceans (Watson, 2001).

The upward bias of the catch data is confirmed by the *FiB* index, which shoots up in the mid 1990s (Fig. 6, center), and more indirectly by the "zero growth policy" promulgated by the Chinese government itself, which decreed in 1999 that catches should not increase beyond their 1998 level (see Fig. 6, top open circles; Pang and Pauly, 2001). On the other hand, the *FiB* index for Chinese waters remains relatively stable through the 1970s, 1980s, and early 1990s, allowing estimation of a transfer efficiency of 6.2% (Fig. 6, bottom). This value is nicely bracketed by the previous two estimates of 6.8% and 5.9%, hence confirming that there is no need to assume the marine ecosystems along the coast of China function in a manner fundamentally different from other shelf ecosystems (Watson, 2001; Watson and Pauly, 2001).

Conclusions

The case studies presented above confirm that the "fishing down" phenomenon, initially presented in Pauly et al. (1998a) and based on an analysis of non-disaggregated data by FAO statistical area, is ubiquitous and much stronger than previously believed. Particularly, it turns out that the decline of mean TL is underestimated when the underlying catch data are taxonomically and/or geographically over-aggregated. The former of these two factors has been discussed previously (Pauly and Palomares, 2000, 2001), but not the latter. However, in retrospect, it is obvious that geographic over-aggregation should have a strong effect. It is thus important, when using mean TL as an indicator of ecosystem status, to use data that are sufficiently disaggregated in geographic terms to correspond to well-defined ecosystems.

The method presented here for estimation of transfer efficiency from fisheries catches and mean TL will require further investigation. We are encouraged, however, by the consistency of our preliminary estimates of TE, and their closeness to values estimated earlier using a more rigorous approach (Christensen and Pauly, 1993), which lead to a global average of 10% for marine ecosystems in general (Pauly and Christensen, 1995).

Moreover, we are intrigued by the possibility of applying this approach to derive spatial maps of ecosystem transfer efficiencies as can be achieved for entire ocean basins using time series of spatial catch maps such as documented for the period from 1950 to the late 1990s, and for ½° × ½° cells by Watson and Pauly (2001), and which are easy to complement by matching series of TL maps. This would have to consider, however, that the plots of the *FiB* index derived for the various cells of those maps, and used to identify "flat" periods, have to be recomputed iteratively, once a first
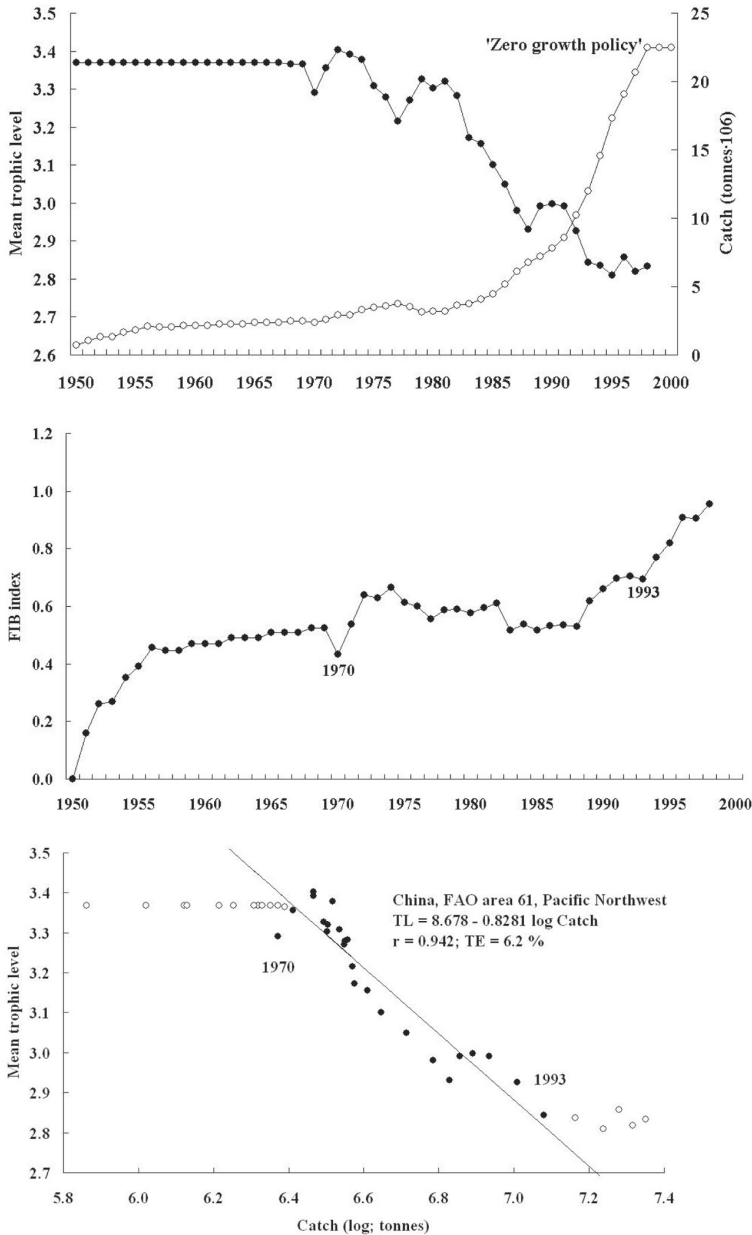
Figure 6. Analysis of the Chinese marine fisheries catch statistics submitted to FAO (FAO area 81; northwestern Pacific). Top: reported catches (open circles), which had long stagnated (and whose crude taxonomic resolution remained constant from 1950–1969, hence the absence of a TL trend (solid circles) during that period; established as for Fig. 4), increased rapidly in the later 1980s, and especially in the 1990s, so much that the Chinese government decreed, in 1999, that these reports should not increase beyond their level in 1998; these data, although unreliable and taxonomically over-aggregated, indicate a strong impact of fishing on mean TL. Center: the *FiB* trends computed from these data confirms that the reported increase for the period from the mid to the late 1990s is incompatible with fisheries operating in the same area (ecosystem), and thus support the inference that over-reporting occurred. Bottom: the more or less stable period from 1970–1993 allow estimation of TE = 6.2%.

estimate of TE has been obtained (based on an input value of, say TE = 10%, as used here), until input TE ≈ output TE.

Overall, we believe that achieving sustainability in fisheries will have to include more or less stable catch compositions, indicated by "flat" series of mean TL, themselves indicative of stable (relative) abundances in the underlying ecosystems. This would imply the use of mean TL and/or of related statistics as indicators of sustainability.

Acknowledgements

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