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Approaches for dealing with three sources of bias when studying the fishing down marine food web phenomenon

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In the Mediterranean as elsewhere, fisheries assessments are performed on a single-species basis, although the fisheries that are to be regulated tend to be multispecies in nature, and to have broad impacts on the ecosystems in which these species are embedded.

These impacts of fisheries have led, in the late 1990s, to a growing interest in “ecosystem-based management” (see *e.g.* NRC, 1999). Moreover, though the full implications of such form of management continue to remain elusive, there is a growing understanding that ecosystem-based management must consider and maintain the structuring role of large predators in marine food webs. The demonstration by Pauly *et al.* (1998a) that the mean trophic level of fisheries landings has been declining in most parts of the world thus confirmed the widespread view that most of the world fisheries are not sustainable in that they increasingly rely on organisms with low trophic levels, such as small fishes and invertebrates, *i.e.*, on the prey species of larger fishes and marine mammals.

The approach of Pauly *et al.* (1998a) having been criticized on methodological grounds, with implicit reference to the Mediterranean (Caddy *et al.*, 1998), this contribution addresses three of the four sources of bias alleged to have affected its results:

- 1- ontogenic effects, due to changes in trophic levels with age, *i.e.*, with size;
- 2- taxonomic effect, due to over-aggregation of landing statistics;
- 3- bottom-up effects, *i.e.*, trophic level changes due to changes in primary productivity.

A fourth potential source of bias, *i.e.*, using the abundance of species groups in landings as proxy for their relative abundances in the ecosystems, was addressed in a first response by Pauly *et al.* (1998b).

Before we can address items (1) to (3), however, we must briefly review the entities called “trophic levels”, crucial to the considerations below.

Trophic levels (TL) express the position of organisms within the food webs that largely define aquatic ecosystems. Their values are set as 1 in plants and detritus, 2 in herbivores (first-level consumers), 3 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:

$$TL_i = \sum_j TL_j \cdot DC_{ij} \quad 1)$$

where TL_j is the fractional trophic levels of the preys 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).

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, 1992; Pauly *et al.*, 2000b), and the ratios of ^{15}N to ^{14}N (Kline and Pauly, 1998; Polunin and Pinnegar, this volume). Fractional TL also allow the development of new approaches to the analysis of aquatic food webs, notably the accurate estimation of the “Primary Production Required” to maintain fisheries (Pauly and Christensen, 1995; Tudela, this volume) and the construction of a series of mean TL values of fish and invertebrates landed by fisheries as a means of evaluating their impacts on marine ecosystems (Christensen, 1998; Pauly *et al.*, 2000b; Stergiou and Koulouris, this volume) thus complementing more traditional approaches (Parsons, 1996). Herein, mean trophic levels (mTL) are computed, for any year y from:

$$mTL_y = \sum_i (TL_i \cdot Catch_{iy}) / \sum_i Catch_{iy} \quad 2)$$

where i refers to the species (groups) included in fisheries statistics (i.e., landings, or preferably catches consisting of landings and discards). This contribution shows how the accuracy of such analyses can be improved.

MATERIAL AND METHODS

Overall approach

Of the three sources of bias mentioned above, the first can be tackled by quantifying the relationship between body size and TL in a wide range of fishes. This can be achieved using data in FishBase (Fig. 1), and using a model of the form:

$$TL_i = 3 + b_i \cdot \log_{10}(\bar{L}_i) \quad 3)$$

where TL_i is the trophic level of species i , \bar{L}_i the corresponding mean length (in cm), 3 is the intercept [for 1 cm larvae, using $\log(1) = 0$, and assuming the larvae feed on herbivorous zooplankton], and b_i is the (mean) slope linking the available $TL_i/\log(\bar{L}_i)$ data pair(s) to the intercept.

This model can then be incorporated into either of two newly-derived equations for estimation of population-weighted estimates of mean TL, presented and illustrated below through an application to *Merluccius merluccius*. The growth parameters $L_\infty = 65$ cm (total length); $K = 0.10$ year⁻¹; $t_0 = -0.17$ year were used for this, based on data from the Northern Aegean Sea for the period 1991-1993 (Stergiou *et al.*, 1997). Also, we used a value of $M = 0.15$ year⁻¹, referring to the Korinthiakos Gulf and the period 1983-1985 (from Papaconstantinou *et al.*, 1985).

The effect of taxonomic over-aggregation (item 2 above) was documented by deriving a series of mean trophic levels using disaggregated data, then comparing the resulting series with one based on the same landing statistics, but in which related taxa had been aggregated before computation of mean trophic level.

Finally, we used the FIB index (Pauly *et al.*, 2000b) to identify bottom up effects (item 3 above), i.e., changes in absolute catches from lower trophic levels. The FIB-index takes the form:

$$FIB = \log [(\sum catch_{iy} \cdot 10^{TL_i}) / (\sum catch_{i0} \cdot 10^{TL_i})] \quad 4)$$

where i is a given species (group) and y a given year, with year “0”, i.e., the year at the start of a series, which serves as anchor. The FIB index changes its value only when a decrease in TL is not matched by a corresponding increase in catch, and conversely for increasing TL. Here, “corresponding” is defined as a 10-fold increase for a decline of one trophic level, as implied by a 10% transfer rates between trophic levels (Pauly and Christensen, 1995).

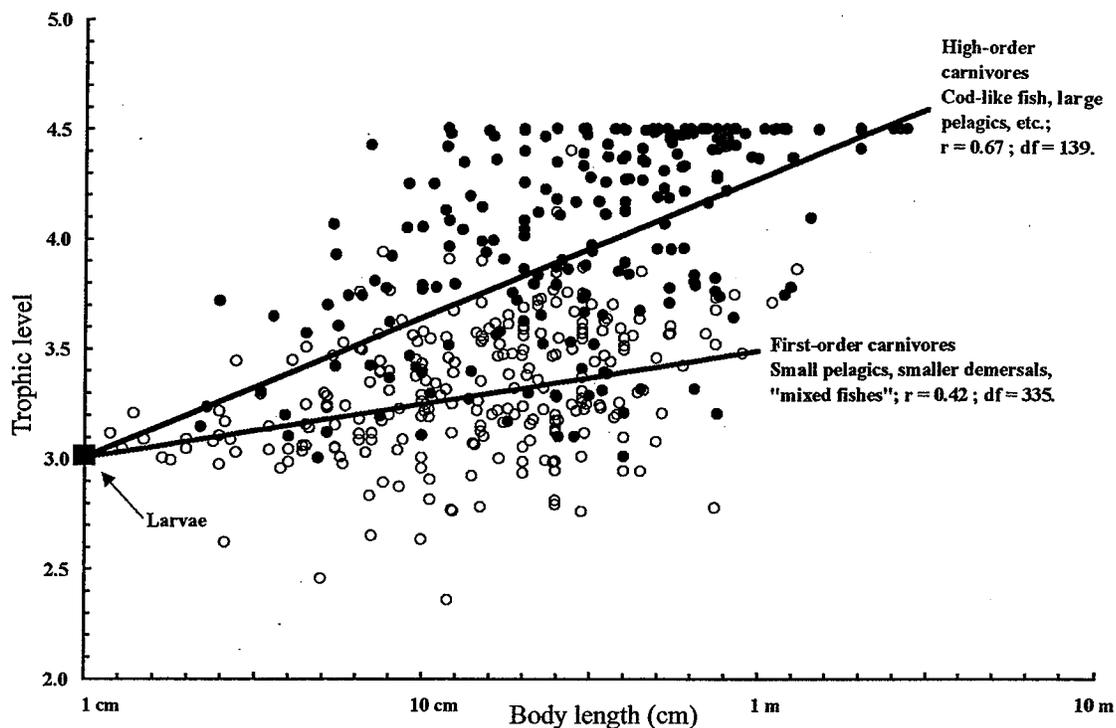


Fig. 1. Plot of estimates of trophic levels (based on diet composition data) vs. body size in fishes, based on data in FishBase (www.fishbase.org). Note steeper slope for high-order carnivores (which include *Merluccius merluccius*), implying a stronger effect of fishing on the mean (population) trophic level (see text).

Length-based relationship between fishing mortality and trophic level

Total mortality (Z), in fish population whose individuals grow according to the von Bertalanffy Growth Function (VBGF), can be expressed by:

$$Z = [K \cdot (L_{\infty} - \bar{L})] / (\bar{L} - L_c) \quad (5)$$

where L_{∞} is the asymptotic length, i.e., the mean size the individuals in the population would reach if they were to live and grow indefinitely, K the rate at which L_{∞} is approached, and \bar{L} is the mean length in the population, computed from L_c upward (Beverton and Holt, 1956). Here, L_c represents the length at entry into the fishery, assuming knife-edge selection. From this, one can derive:

$$\bar{L} = [L_{\infty} + (L_c \cdot Z/K)] / [(Z/K) + 1] \quad (6)$$

Defining exploitation rate (E) as $E = F/Z$, and $E = 1 - (M/K) / (Z/K)$, we also have:

$$\bar{L} = \{ L_{\infty} + [(M/K) / (1-E)] \cdot L_c \} / \{ [(M/K) / (1-E)] + 1 \} \quad (7)$$

for $0 \leq E < 1$. Thus, given Eq. 3, the change in TL (ΔTL) in a fish population where body sizes have been reduced by fishing can be estimated from:

$$\Delta TL = b \cdot \log_{10} \left[\frac{L_{\infty} + \left(\frac{M/K}{1-E} \cdot L_c \right) / \left(\frac{M/K}{1-E} + 1 \right)}{\left(L_{\infty} (M/K \cdot L_c) / (M/K + 1) \right)} \right] \quad (8)$$

where all parameters are as defined above.

The estimates of ΔTL thus obtained are approximate because the TL values corresponding to the mean length obtained from Eq. 8 can only approximate the mean TL value obtained by integrating TL within an age-structured population, our next topic.

Age-based relationship between fishing mortality and trophic level

The mean TL of an age-structured population in which TL changes linearly with (log) length can also be estimated from:

$$TL = \frac{\left(\int_{t_r}^{t_c} N1_t \cdot A \, dt + \int_{t_c}^{t_{max}} N2_t \cdot N3_t \cdot A \, dt \right)}{\left(\int_{t_r}^{t_c} N1_t \, dt + \int_{t_c}^{t_{max}} N2_t \cdot N3_t \, dt \right)} \quad 9)$$

where $N1_t = e^{-M(t-t_r)}$; $N2_t = e^{-M(t-t_r)}$; $N3_t = e^{-F+M(t-t_c)}$; $A = 3 + b \cdot \log_{10}L_t$ (see Eq. 3); $L_t = L_{\infty} \cdot (1 - e^{-K(t-t_0)})$, i.e., the VBGF for growth in length with parameters L_{∞} and K defined as above, and t_0 setting the origin of the VBGF; t_r = the recruitment age, i.e., the age from which on Eq. 9 applies; t_c = the age at entry into the exploited stock, assuming knife-edge selection (i.e., corresponding to L_c ; see above); and t_{max} = the maximum age in the population.

RESULTS AND DISCUSSION

The nomogram in Fig. 2 illustrates three points: (a) that the absolute value of the correction (ΔTL) due to size changes can be substantial, especially when fishing mortality is high; (b) that the sign of this correction is negative, and hence that not considering it underestimates the strength of the fishing down marine food web effects (hence invalidating that part of the critique by Caddy *et al.*, 1998); and (c) that the length-based model (Eq. 8) slightly underestimates the required correction (ΔTL) implied by the age-structured model (Eq. 9).

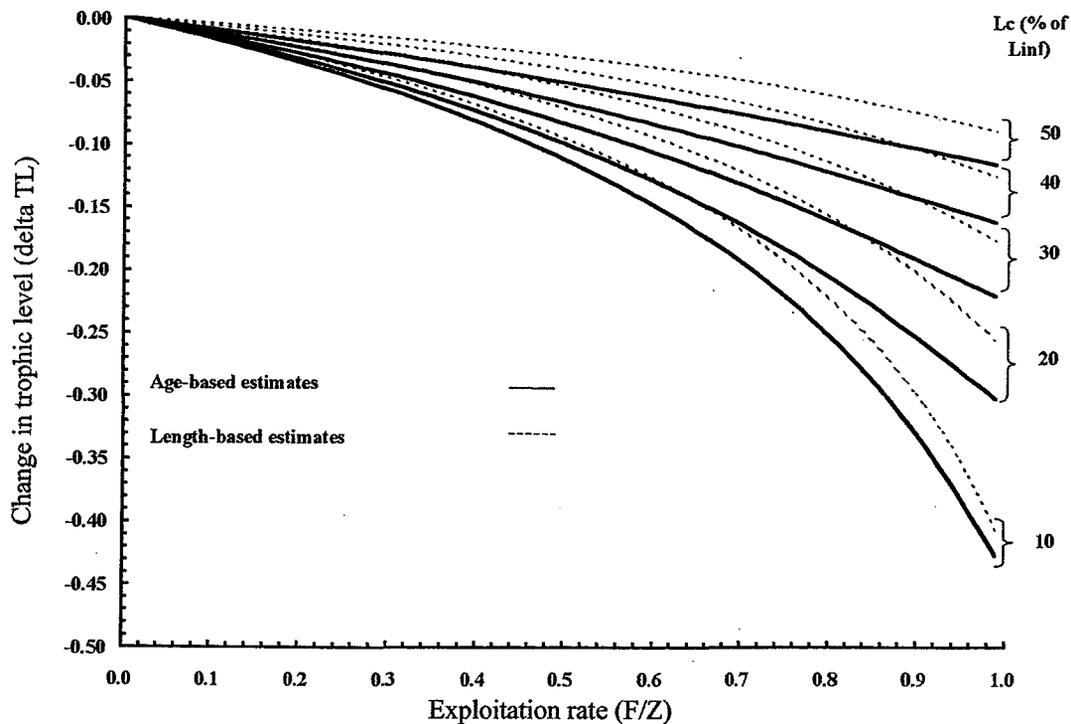


Fig. 2. Nomogram summarizing the impact of fishing (expressed by the exploitation rate F/Z) on the trophic level of high-level predators (as defined in Fig. 1). Note that Eqs. 8 and 9 give similar results, with the length-based model (Eq. 8) slightly underestimating the strength of this effect (see text).

Given these, we propose a generalized model for calculating mTL from multispecies catch (or landings) data, resulting from combining Eq. 2 and Eq. 3, viz:

$$mTL_y = \sum_i (3 + b_i \cdot \log_{10}(\bar{L}_{iy}) \cdot Catch_{iy}) / \sum_i Catch_{iy} \quad 10)$$

where the TL_{iy} for a species i and year y are obtained via a constant, species-specific previous estimate of b_i (as can be derived from pairs of TL_i/\bar{L}_i , and from year-specific mean lengths, obtained either from field data, or via Eq. 7 (but preferably not a mixture of the two length types in the same species)).

As such, Eq. 10 will allow time series of mTL to account simultaneously for both between-species effects (via changes in the species mix, as expressed by the catches) and within-species effects (via the impact of mean length changes on the TL estimates).

Moreover, Eq. 10 should also be useful for analyses such as that of Harmelin-Vivien (this volume), where relative abundances in the ecosystems (*e.g.*, in comparisons of fished and unfished areas) are used instead of catches or landings. Also note that species-specific estimates of b_i , required to convert mean length into TL estimates, can be derived from data in FishBase for most of the commercial species of the Mediterranean (Froese and Pauly, 1999), with more species to be included in forthcoming updates (see: www.fishbase.org).

Fig. 3 documents the effect of over-aggregating fisheries statistics on the estimation of mTL. As might be seen, the downward trend of mTL estimated from detailed catch statistics (over 80 % of catch reported at species level) disappears when these same data are regrouped into coarser entities, here the functional groups of FAO's International Standard Classification of Aquatic Animals and Plants (ISCCAAP). As only 50 % of the world's, and about 65 % of Mediterranean landings are reported on a per-species basis, the "fishing down" effect reported by Pauly *et al.* (1998a) for the world as a whole, and for the Mediterranean (see Fig. 1 in Executive Summary, this volume) will underestimate the decline of mTL values. Note that this effect is added to the ontogenic effects discussed above, thus further invalidating the critique of Caddy *et al.* (1998).

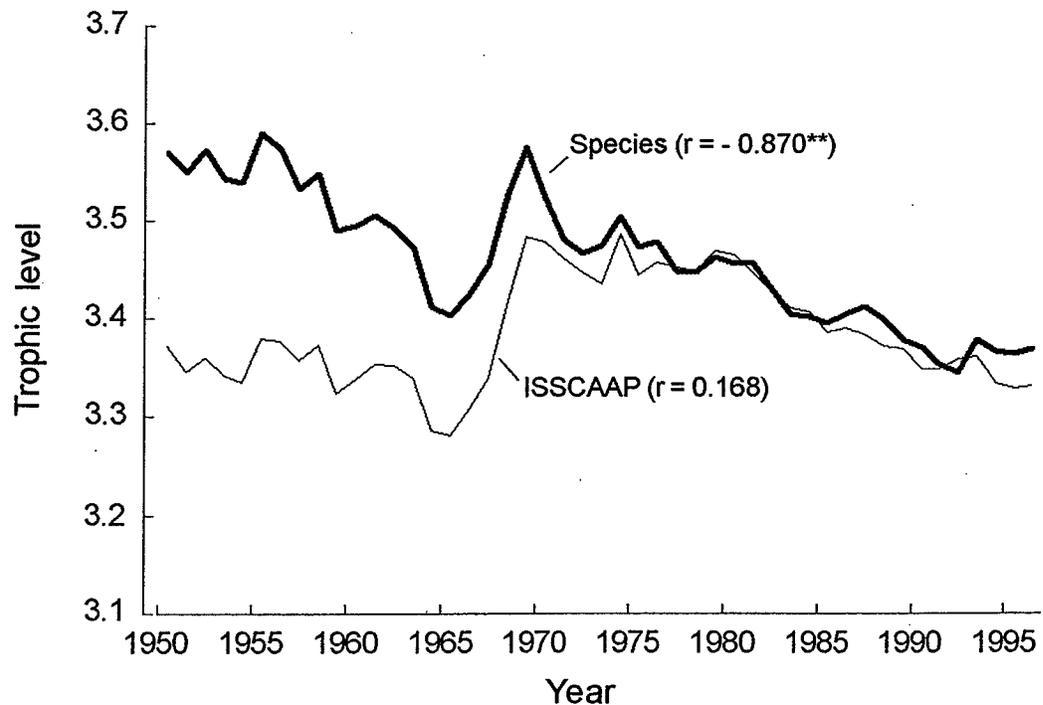


Fig. 3. Trends of mean trophic level in the Eastern North Atlantic (FAO Area 27), without (thick line) and after (thin line) aggregating the catch data into the broader ISSCAAP categories. As may be seen, the resulting loss of taxonomic information leads to a disappearance of the previously significant downward trend of mean trophic level (see text).

Fig. 4 finally contrasts time series of the FIB index for two FAO areas with strongly diverging features: the Northwest Atlantic, where TL and catches increased from the 1950s to the 1970s, following a relative increase of effort on previously under-exploited deep demersal stocks, which

eventually collapsed (as in many other parts of the world), and the Mediterranean, where catches increased without a strong geographic expansion of the fisheries, suggesting a bottom-up effect (eutrophication leading to increased catches of herbivores, *e.g.*, bivalves and small pelagics; see Caddy, 1993).

This may justify the notion that the Mediterranean is somehow “different”, though its food webs may also end up being “fished down.”

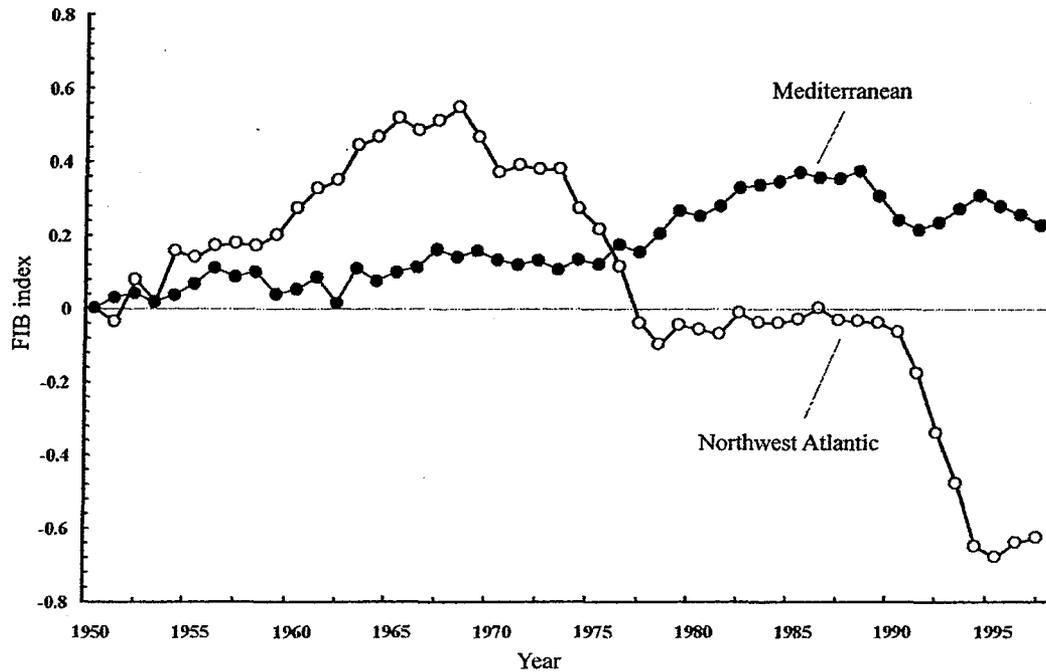


Fig. 4. Simultaneously considering catch variations and changes in trophic levels, as allowed by the FIB-index, shows that in the Mediterranean, catches increased as trophic levels declined, suggesting a bottom-up effect (i.e., eutrophication possibly leading to increased populations and catches of bivalves and small pelagics). The Northwest Atlantic, in contrast, saw “extensions” (new areas and new species fished), from 1950 to the mid 1960s, then a stepwise decline, culminating in the 1990s in a collapse of large demersal fishes with high trophic levels (especially Atlantic cod), but without compensating effects from the bottom-up.