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Combining indicator trends to assess ongoing changes in exploited fish communities: diagnostic of communities off the coasts of France

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We present a method for combining individual indicator results into a comprehensive diagnostic of fishing impacts on fish populations and communities. A conceptual framework for interpreting combined trends in a set of simple indicators is proposed, relying beforehand on qualitative expectations anchored in ecological theory. The initial state of the community is first assessed using published information. Which combinations of trends are acceptable or undesirable is decided, depending on the initial status. The indicators are then calculated from a time-series and their time trends are estimated as the slopes of linear models. Finally, the test results are combined within the predefined framework, providing a diagnostic on the dynamics of fishing impacts on populations and communities. The method is demonstrated for nine coastal and shelf-sea fish communities monitored by French surveys. Most communities were persistently or increasingly impacted by fishing. In addition, climate change seems to have contributed to changes in East Atlantic communities.

Cet article propose une méthode utilisant des indicateurs pour élaborer un diagnostic sur les effets de la pêche sur les populations et les peuplements de poissons. Un cadre conceptuel permet d'interpréter les tendances conjointes d'indicateurs à partir de la théorie écologique. L'état initial du peuplement est d'abord évalué sur la base d'informations publiées. En fonction de l'état initial et des objectifs de gestion, les combinaisons des tendances sont qualifiées d'indésirables ou satisfaisantes. Les indicateurs sont ensuite estimés à partir de données de campagnes de pêches scientifiques: abondance et longueur moyenne d'une sélection de populations, nombre, biomasse totale, poids moyen et longueur moyenne dans le peuplement, et la pente du spectre de taille multispécifique. Les tendances temporelles de ces indicateurs sont testées au moyen d'un modèle linéaire, et les résultats des tests sont combinés en un diagnostic final. La méthode est mise en œuvre pour neuf peuplements de poissons côtiers et du plateau continental, suivis par des campagnes françaises. Il en résulte que la plupart de ces peuplements sont affectés par la pêche de manière stationnaire ou croissante. Par ailleurs, les changements climatiques dans l'Atlantique Nord-Est contribuent aussi à des modifications dans les peuplements de poissons de cette région.

Keywords: ecosystem approach to fisheries management, indicators, multispecies fisheries, scientific surveys.

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Introduction

There is increasing interest in ecosystem-based fisheries management, from scientists (Anon., 1999; ICES, 2000), fisheries management bodies, and other stakeholders (FAO, 2001; Council of the European Union, 2002). Within this approach, it has been widely recognized that management goals and aspirations have to be translated into operational objectives, which can be tracked by indicators (FAO, 2003; Degnbol and Jarre, 2004). Comparison of indicators with target and limit reference points provides decision criteria and also measures how well management performs. However, whereas reference points are rather well defined in single-stock assessments, an ecosystem approach complicates matters for several reasons. First, several indicators are needed to address the many dimensions of an exploited ecosystem. This makes an indicator-based assessment multivariate in essence (Link, 2005). If every single indicator could be associated with a reference point, this would give many decision criteria that would not necessarily be consistent and would have to be reconciled by managers. To address this, multivariate reference spaces can be mapped in the indicator space from the analysis of long-term multivariate data series with contrasts among exploitation periods (Link *et al.*, 2002) or regions (Charvet *et al.*, 2000), or from simulations (Pitcher and Preikhost, 2001). Second, even for a single objective such as maintaining the size structure of a fish community, several indicators may be useful to help interpret the information they convey (Shin *et al.*, 2005), because whereas indicators that are sensitive to fishing are numerous, none of them is specific (Rochet and Trenkel, 2003). That is, each of them varies in response to a number of environmental factors. Because indicators might differ in their sensitivity to different factors, combining several of them might help disentangle the effects of fishing from those of other factors. Third, reference points might be technically and theoretically difficult to develop and justify (Rochet and Trenkel, 2003; Jennings and Dulvy, 2005) owing to difficulties in converting broad objectives into specific limits and targets, insufficient data series at appropriate scales, and unavoidable bias introduced by sampling gears. Instead, Jennings and Dulvy (2005) suggested that time trends in indicators could be compared with reference directions to judge management performance by evaluating whether ecosystem status is improving or deteriorating. This is achievable because, for relevant indicators of the impact of fishing on a community, reference directions are well established: we know whether fishing will increase or decrease the value of the indicator (Rochet and Trenkel, 2003). Fourth, the issue that reference points should incorporate medium- to long-term variability in the environment (Lassen, 1999), in life history (Rahikainen and Stephenson, 2004), or in prey availability and in predation mortality (Collie and Gislason, 2001), has been raised at the single-stock level. This would be amplified in an ecosystem approach, owing to the many dimensions to be considered, and the many interacting factors affecting each dimension: multivariate reference spaces would be conditional on the set of environmental conditions that prevailed during the period of the data series, or on the assumptions used in the simulations. Moreover, a reference space would be moving in an unpredictable manner when environmental conditions or the societal background change. At a given time, it seems more feasible to establish whether an ecosystem is moving in a desirable direction, rather than to determine how far it is from a broadly specified, moving multidimensional target (or limit). This statement is still paradoxical: a desirable direction can only be defined in comparison with a known target. The solution of this paradox lies both in "at a given time" and "broadly specified". Whereas it seems unreasonable to determine a reference region in the space of indicators relevant to both management objectives and the current situation, it might be possible to assess whether a situation was satisfactory or not some years later, using additional information, that was not available or not interpretable at the time.

We propose a method to combine simultaneous trends in several indicators into a diagnostic of the dynamics of a fish community. The first step consists of assessing the initial status of the community relative to its desirable state. If a community was considered to start in a non-impacted state, the absence of time trends (stationarity) is acceptable. However, if a community was already impacted, no change is not good news. A trend-based assessment will have to answer two questions, based on different sources of information: (i) what was the status of the community at the beginning of the assessment period, and (ii) has the condition of the community improved or deteriorated since then? To achieve this, a conceptual framework for interpreting combined trends in a set of simple indicators of fishing impacts on fish populations and communities is proposed, relying beforehand on qualitative expectations anchored in ecological theory. Which combinations are acceptable or undesirable can then be decided, depending on the initial status. These indicators are then calculated from a data time-series, and their time trends are estimated as the slopes of linear models. Finally, the test results are combined within the predefined framework, providing a diagnostic on the dynamics of fishing impacts on populations and communities.

Moving from single-stock towards ecosystem-based management means taking note that exploitation may not only modify target populations, but whole communities, i.e. all species that directly interact with the target species through competition, predation, other biological proceses, or that are incidentally taken as bycatch (Hall, 1999). Integrative community indicators are not yet well developed and tested: it seems reasonable to complement them by monitoring a wide selection of populations, including both target and non-target species (Hall and Mainprize, 2004). This is the reason why our assessment targets two levels: populations and the community as a whole. To address the impacts on both commercial and bycatch species in a comparable way, these indicators are estimated from scientific survey data.

Scientific trawl surveys have been conducted in the seas around France covering nine coastal and shelf-sea communities, with the primary purpose of providing abundance or recruitment indices for stock assessments. Recently, these objectives have been broadened towards a more holistic approach to fisheries assessment, with an increasing proportion of taxa being identified, weighed, and measured. The primary purpose of this study is to implement, on a large scale, an assessment of the impact of fishing on these fish communities. Combining analyses across exploited ecosystems provides an empirical test of the approach we propose. The price to pay for this comparative approach is that the set of indicators had to be restricted to those that could be estimated by similar methods across all ecosystems.

Material and methods

We selected six continental shelf communities supporting mixed fisheries, which are monitored by bottom-trawl surveys (Figure 1, Table 1). We also assessed three estuarine communities identified as nursery areas for commercially important stocks exploited elsewhere in mixed fisheries; these communities are sampled by beam-trawl surveys. The main purpose of the bottom-trawl surveys was initially to provide abundance indices of commercial stocks, whereas the coastal beam-trawl surveys were designed to provide recruitment indices. In all surveys, all fish are identified and counted, and most or all are measured. Hence, these surveys provide a picture of the total fish community, at least in the most recent period. In all surveys, the sampling design is stratified according to depth and some other criterion (e.g. North/South in the Bay of Biscay, bottom substratum in the Vilaine Estuary).

Survey trawls do not sample all species equally well. Some species are rare or aggregated in the survey area; some have part of their range outside the survey area; others escape from the gear for various reasons. This contributes to sampling biases and large sampling variances. Population indicators were estimated only for the fish species for which a reasonable precision could be achieved (Table 1). The criteria for selecting a species were a sufficient occurrence (proportion of hauls with the species

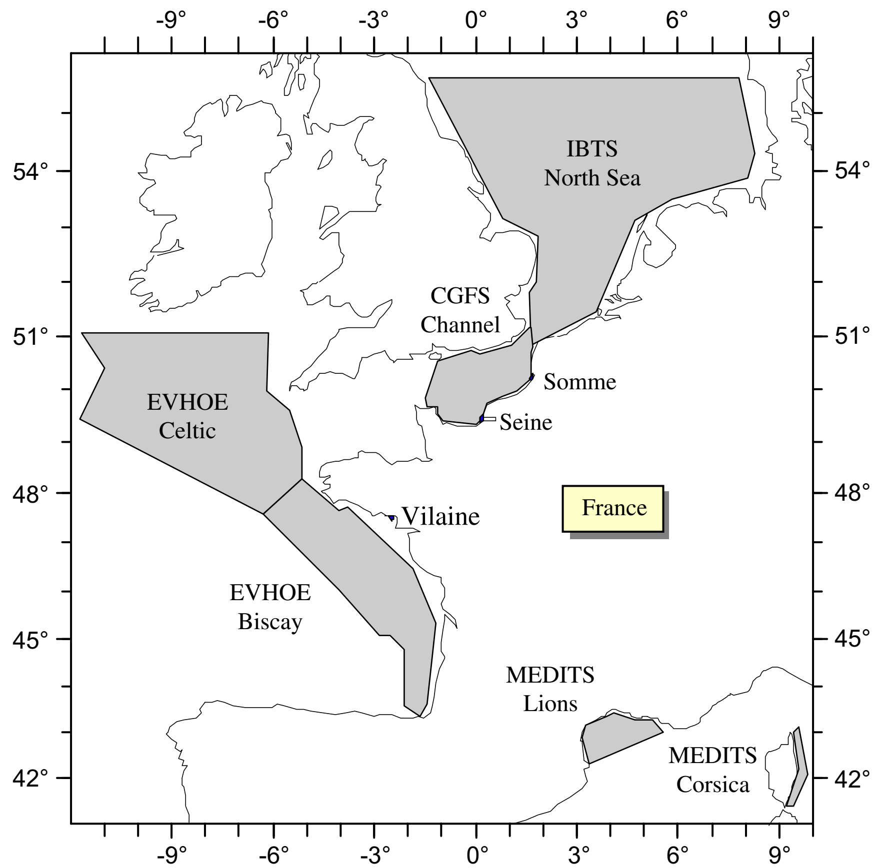

Figure 1. Map of communities and survey areas (see details in Table 1).

Table 1. List of the survey data used in the analysis.

Community	Survey	Time-series available	Season	Number of strata	Number of hauls per year	Total area covered (km^2)	Depth range (m)	Number of species caught	Number of species included in population analysis
Seine Estuary	Seine	1995–2002	Autumn	12	45	550	0–20	56	9
Somme Estuary	Somme	1999–2002	Autumn	4	48–54	718	0–20	41	9
Vilaine Estuary	Vilaine	1982–2002	Autumn	4	19–46	330	0–20	37	11
East Corsica	MEDITS*	1995–2001	Spring	2	13–25	4 562	10–800	163	22
Gulf of Lions	MEDITS*	1995–2002	Spring	2	64–76	13 860	10–800	179	22
Southern North Sea	IBTS†	1990–2000	Winter	80	143–210	252 813	15–100	106	13
Eastern Channel	CGFS‡	1997–2002	Autumn	17	83–109	30 672	7–82	76	18
Celtic Sea	EVHOE§	1997–2002	Autumn	12	53–82	150 000	20–400	103	43
Bay of Biscay	EVHOE§	1987–2002	Autumn	14	56–113	72 500	20–600	194	51

*International bottom trawl surveys in the Mediterranean (Anony., 1998).
†International Bottom Trawl Survey (ICES, 1996).
‡Channel Ground Fish Survey (Carpentier *et al.*, 1989).
§EValuation des ressources Halieutiques de l'Ouest de l'Europe par campagnes de chalutages programmés (ICES, 1991b).

present being >5%), a sufficient density and/or commercial interest, and availability of length data (e.g. in the MEDITS surveys, only a subset of species is measured). The species used for the estimation of population and community indicators are reported at http://www.ifremer.fr/emh/publications.htm, and in Rochet et al. (2005).

Indicators

Seven indicators for measuring fishing impacts on the population and community levels were selected (Table 2). Two indicators of population state, log abundance, and average length in the population, were calculated for a large set of selected species (see previous section). Both are expected to decrease in exploited stocks (Beverton and Holt, 1957), so the direction to watch out for is a declining trend. The same interpretation of trends was assumed to apply to non-target species, most of which are taken as bycatch and hence should suffer similar impacts. Some non-target species might increase in abundance because of lower predation by depleted target predator species, or in average length because of reduced competition (Hall, 1999). However, it was assumed that for a majority of species, these indirect effects of fishing would be lower in magnitude than the direct effects.

Similar direct effects of fishing are expected at the community level. Total community biomass and number might decrease under strong fishing pressure, and this decrease would impair the productivity of the dependent fisheries (Rochet and Trenkel, 2003). Average length and weight (all species) decrease in exploited communities (Jennings *et al.*, 1999; ICES, 2001). The slope and the intercept of the size spectrum were also estimated. Fishing is generally expected to decrease the slope of the size spectrum (Pope and Knights, 1982; Gislason and Rice, 1998). In order to consider only the descending limb of the spectrum, size spectra were calculated for all sizes above the smallest length class considered to be fully caught by the sampling gear (15 cm for most surveys). Size spectrum slope and intercept were not estimated for the coastal communities, where surveys target juvenile fish <15 cm. In these communities, the variability in size spectra characteristics is

Table 2. Selected population and community indicators. For further details, see Trenkel and Rochet (2003).

Level	Indicator	Definition	Expected effect of fishing
Population	$\ln(\hat{N}_{i,t})$	Ln-transformed population abundance for species i	Decrease
	$\bar{L}_{i,t}$	Average length of population i	Decrease
Community	$\hat{B}_t$	Total biomass	Decrease
	$\hat{N}_t$	Total abundance in the community	Decrease
	$\bar{W}_t$*	Average weight	Decrease
	$\bar{L}_t$	Average length	Decrease
	β_t†	Size spectrum slope	Decrease
	α_t†	Size spectrum intercept	Unknown

*Average weight was estimated for the overall community as the ratio of total biomass to total abundance, as individual weights were not measured.
†ln $N_l(t) = \alpha_t \exp(\beta_t(l - \bar{l}_t))$, where N_l is number at length l and $\bar{l}_t$ is medium length in the community, year t, and the parameters are estimated using a Generalized Linear Model with length classes weighted by the number of hauls in which the length class was present. Note: β_t is generally a negative number, hence decreasing β_t implies a steeper (more negative) slope.

mainly driven by fluctuations in the recruitment of the dominant species, which are influenced both by interannual environmental variability and by survey timing, and might not reflect population-level changes in abundance.

Having estimated an indicator I for a given data series, the parameters of a linear time trend in the indicator $\hat{I}_t = a + bt$ were estimated, and the null hypothesis that $b = 0$ was tested (for details see Trenkel and Rochet, 2003). Two-sided tests were preferred to one-sided tests with the alternative hypothesis that the detected change is due to fishing. This is because trends in the direction opposite to fishing impacts also provide a signal, although with a different meaning (see below). Linear regression analysis assumes that indicators are normally distributed. This should be the case for all indicators that are mean values of some random variable as a result of the central limit theorem. This might also be expected to be true for total abundance or biomass in the community. Finally, ln-transformed abundance indices are commonly normally distributed. Hence, the fundamental conditions for simple linear regression can be expected to be fulfilled for all indicators used in this study. Instead of linear regression, non-parametric tests such as the Mann−Kendall test could have been used. The advantage of the linear regression method is that in addition to providing a test for change, the slope estimates allow ordering of species by intensity of trend.

From indicators to community diagnostics

We first performed a quick assessment of the status of each community at the start of the data series. Signs of fishing impacts were gathered, based on stock assessments by advisory bodies, published evidence, and any other available information. We used the following criteria: communities with several overexploited commercial stocks or bearing a high fishing effort or destructive fishing methods (discarding, habitat damaging, etc.) were decided impacted; non-impacted communities were those with none of these characteristics.

We defined a framework for interpreting combinations of pairs of indicators at the population and community levels by setting up diagnostic tables (Table 3). For this first application of the method, interpretations were based on common sense and basic ecological theory (Beverton and Holt, 1957; Hart and Reynolds, 2002; Sibly *et al.*, 2003). For example, if average length of fish in a population increased whereas its abundance remained stationary, this means that there were both more large fish and fewer small ones (Table 3a). This could be due either to reduced mortality (fishing or natural mortality, or both) concomitant with decreased recruitment, or to faster growth, which might in turn result from changes in the environment (food, temperature, competition…), or from a genetic change in growth rate. Changes in recruitment could be due to changes in reproduction (lower reproductive capacity of the stock, e.g. fishing-induced change in age-structure, maturity, fecundity; disturbance of spawner aggregations…), changes in larval survival (change in egg size, in growth or mortality rates, attributable to changing environmental conditions: food/predation or larval drift patterns altered by ocean circulation). More or less fish in the survey area might also indicate a shift in the distribution area of the population, because of changes in environmental conditions, in migration patterns, or in the spatial distribution of fishing effort. Each cell in Table 3 contains a list of potential mechanisms, which could be further disentangled by the use of additional indicators (see Discussion). A similar framework was developed for community indicators and combined in a two-dimensional table to facilitate presentation (Table 3b). Total abundance and biomass of the community were selected as the two entries of the table, but other pairs could have been used as well.

Which indicator trend combinations were acceptable was decided on the basis of interpretation tables and depended on the initial state assessment. Based on a precautionary principle, we qualified "deteriorating" (shaded cells in Table 4a) as any combination of trends in population indicators for which one of the potential mechanisms was increased fishing mortality, or the situation requires reducing fishing pressure to reverse the trends, even if not caused by fishing. When the community was considered impacted at the beginning of the time-series, situations where none or only one indicator was trending in the direction opposite to expected fishing impact were qualified as "not improving" (hatched cells in Table 5a).

A formal test procedure was developed to assess trends in the two indicators over all populations jointly. Probabilities for each trend combination were estimated, based on the null hypothesis that populations were stable and that trends in log abundance and average length are independent. Under a neutral model, a community is made up of independent individuals with similar vital rates (Bell, 2001). Hence, populations (groups of individuals) might increase or decrease in abundance or average length independently by chance. Under this null hypothesis, performing independent tests on the two indicators result in the probabilities of combined trends summarized in Tables 4a and 5a. For example, the probability that both log abundance and average length significantly increase in a given population is equal to the product of the risk levels of each test, that is, for two-sided tests $(\alpha/2)^2$, i.e. 0.025^2. A two-step procedure was used, depending on the initial state, to test whether overall there was evidence that populations were moving towards undesirable directions or staying in undesirable states (Tables 4b and 5b).

At the community level more than two trends have to be combined: a sequential procedure was used, which was summarized as a decision tree (Figure 2). A community was said to deteriorate as soon as one indicator was pointing in a deteriorating direction. In addition, combinations of trends that might be interpreted as fishing down the foodweb or decreasing system production (Table 3b) were also labelled undesirable. By contrast, an impacted community was said to recover only if both total abundance and

Table 3. Interpreting trends in indicators jointly. (a) Population indicators: potential mechanisms for each combination of trends in log population size and average length: direct effect of fishing, *environmental effect*, ***a combination of fishing and environment***. (b) Community indicators: potential mechanisms for each combination of trends in total community abundance and biomass. In each cell, the first line gives the mechanism, the bullets suggest potential consistent signals, the last line an interpretation.

(a) $\overline{L}_{i,t}$	$\ln(\hat{N}_{i,t})$		
	Increase	Stationary	Decrease
Increase	• More large fish: mortality decreases (F or M) • *Shift in spatial distribution: more large fish* • ***More fish and faster growth***	• ***Faster growth*** • More large fish (mortality decreases or *distribution shift*) and *decreasing recruitment*	• Less small fish: ***decreasing recruitment*** or more undersized fish killed • *Shift in spatial distribution: less small fish* • ***Less fish of any size and faster growth***
Stationary	• More small and large fish: good recruitment and low mortality (F or M) • *Shift in spatial distribution: more fish in survey area* • More old fish: mortality decreases (F or M) and ***slower growth*** • *More small fish and faster growth*		• Less fish of all sizes: increased mortality (F or M) and poor recruitment • *Shift in spatial distribution: less fish in survey area* • Less old fish and ***faster growth*** • Less small fish: decreasing recruitment and slower growth
Decrease	• More fish and slower growth (density dependence) • More small fish: *increasing recruitment* or improved selectivity (undersized fish not killed) • *Shift in spatial distribution: more small fish*	• *More small fish* and increased mortality (F or M) • ***Slower growth***	• Less large fish: increased mortality (F or M) • *Shift in spatial distribution: less large fish* • ***Less fish and slower growth***

(b) Total biomass $\hat{B}_t$	Total number $\hat{N}_t$		
	Increase	Stationary	Decrease
Increase	More animals • Several populations $\ln(\hat{N}_{i,t})$ increase *System productivity increase* *Decreasing fishing impacts*	Bigger animals • $\overline{W}_t$ increase • Several populations $\overline{L}_{i,t}$ increase *Improved transfer efficiency*	Less and much bigger animals • $\overline{W}_t$ and $\overline{L}_t$ increase • β_t increase • Several populations $\ln(\hat{N}_{i,t})$ decrease • Many populations $\overline{L}_{i,t}$ increase *Decreased inputs to the system (primary production/animal reproduction)*
Stationary	More and lighter animals • $\overline{W}_t$ decreases • β_t decreases • Several populations $\ln(\hat{N}_{i,t})$ increase • Several populations $\overline{L}_{i,t}$ decrease *Fishing down the marine foodweb*	Species replacements Compensations	Less and bigger animals • $\overline{W}_t$ increase • β_t increases • Several populations $\ln(\hat{N}_{i,t})$ decrease • Several populations $\overline{L}_{i,t}$ increase *Decreased inputs to the system (primary production/animal reproduction)*
Decrease	More and much lighter animals • $\overline{W}_t$ and $\overline{L}_t$ decrease • β_t decreases • Several populations $\ln(\hat{N}_{i,t})$ increase • Many populations $\overline{L}_{i,t}$ decrease *Fishing down the marine foodweb*	Lighter animals • $\overline{W}_t$ decreases • Several populations $\overline{L}_{i,t}$ decrease *Fishing down the marine foodweb*	Less animals • Several populations $\ln(\hat{N}_{i,t})$ decrease *System productivity decreases* *Fishing on too small animals*

Table 4. Diagnostic table when initial population state is satisfactory. (a) Expected probabilities for combinations of trends in population indicators under the null hypothesis of no change. Cells where one of the potential driving mechanisms is increasing fishing impacts are shaded. Numbers are the expected probability of each cell when individual tests are performed independently with alpha = 0.05 (product of marginal probabilities). The expected probability of the shaded area (undesirable trends) is 0.04875. (b) Formal test procedure to decide whether overall, populations are moving in undesirable directions. Tests are conducted sequentially and the procedure is stopped as soon as a conclusion regarding fishing impact is reached.

(a)	$\ln(\hat{N}_{i,t})$			
$\bar{L}_{i,t}$	Increase	Stationary	Decrease	Total
Increase	0.000625	0.02375	0.000625	0.025
Stationary	0.02375	0.9025	0.02375	0.95
Decrease	0.000625	0.02375	0.000625	0.025
Total	0.025	0.95	0.025	1

(b) Step	*Question asked*	H_0	Test	Conclusion
1	*Did fishing impact appear?*	No trend towards increasing fishing impact (number of populations in shaded area consistent with stable neutral community)	Binomial model: probability of increasing fishing impact p = 0.04875, number of trials = number of populations	If rejected, evidence for increasing fishing impact If accepted, go to 2
2	*Did the populations remain stationary or move in directions different from increasing fishing impacts?*	Stationary populations (number of populations in each cell of Table 4a consistent with stable neutral community)	G-test (log-likelihood ratio test; Sokal and Rohlf, 1995) comparison of expected and observed frequency distribution	If rejected, evidence for change probably not due to fishing. If accepted, populations stable

biomass were increasing (Figure 2b). For example, starting from an impacted state, the trend in average length is considered first. If it is decreasing, the community is said to be deteriorating and the procedure is stopped. If not, the trend in total biomass is examined in the same way, and so on. The indicators estimated with the best precision and giving clues about the interpretation of trends (Table 3b) entered the decision tree first. Some indicators cannot be considered independent of others. For example, as average weight was estimated as the ratio of total biomass to total abundance, a time trend in mean weight can be tested only if both components are stationary. The probability of ending in a given combination of trends under the null model of a stable neutral community can be calculated as the product of the probability of each path.

Results at the population and community levels were finally combined into a final diagnostic using a simple rule: as soon as one level was found to be deteriorating, so was the system. Conversely, improvement at the two levels was necessary to conclude that the system was recovering.

Results

Initial state assessments

Most of the communities examined were already impacted by fishing at the beginning of the survey periods (Table 6). The two exceptions are the shelf to the east of Corsica, where industrial fishing effort is low, and the Vilaine Estuary, where the destructive shrimp fishery declined in the early 1980s (Forest, 1988).

Coastal communities

In both the Seine and Somme Estuaries, no significant trends were found for any of the indicators (Table 7, Figure 3a). These already impacted systems remained impacted (stationary). For the Somme Estuary, however, this result might be due to the very short time-series (4 years only).

In the Vilaine Estuary, three populations (pollack, *Pollachius pollachius*; plaice, *Pleuronectes platessa*; and dab, *Limanda limanda*: $p = 10^{-4}$–0.04) decreased and two (wedge sole, *Dicologoglossa cuneata*; and grey gurnard, *Eutrigla gurnardus*) increased among 11 species analysed. This change is attributed to fishing by the binomial test with a p-level of 0.0128 (3/11 populations in the shaded region in Table 7). On the other hand, no sign of a deterioration was detected at the community level (Figure 3b).

Mediterranean communities

On the shelf to the East of Corsica no significant trends in population abundances were found. Three trends in average

Table 5. Same as Table 4, when initial population state is strongly impacted by fishing. (a) Cells where one of the potential driving mechanisms is increasing fishing impacts are shaded; cells with suspicion of stationary fishing impacts are hatched. The expected probability of the shaded area (undesirable trends) is 0.04875 and the hatched area (no improvement) 0.950625. (b) Formal test procedure.

(a)	$\ln(\hat{N}_{i,t})$			
$\bar{L}_{i,t}$	Increase	Stationary	Decrease	Total
Increase	0.000625	0.02375	0.000625	0.025
Stationary	0.02375	0.9025	0.02375	0.95
Decrease	0.000625	0.02375	0.000625	0.025
Total	0.025	0.95	0.025	1

(b) Step	*Question asked*	H_0	Test	Conclusion
1	*Did the populations remain stationary?*	No change (number of populations in the three types of cell in Table 5a consistent with stable community).	Multinomial model: cell probabilities from Table 5a, number of trials = number of populations	If rejected, go to 2 If accepted, populations remain impacted
2	*Did populations show evidence of increasing fishing impacts, or the opposite?*	No trend towards increasing fishing impact (number of populations in shaded area consistent with stable neutral community).	Binomial model: probability of increasing fishing impact = p= 0.04875, number of trials = number of populations	If rejected, populations deteriorate If accepted, populations improve

length were significant, with hake (*Merluccius merluccius*, $p = 0.0025$) and horse mackerel (*Trachurus trachurus*, $p = 0.04$) decreasing, and thornback ray (*Raja clavata*) increasing ($p = 0.04$, Table 7). Among the community indicators, only the slope of the size spectrum was found to increase, indicating an increase in the proportion of large fish in the community (Figure 3b).

By contrast, the absence of any significant change in the Gulf of Lions is worrisome, as this community was considered already severely impacted at the beginning of the assessment. The thornback ray population was found to decrease ($p = 0.003$, Table 7), and not a single individual was caught in 2002 nor in 2003. Two populations were found to change significantly in average length, red mullet, *Mullus barbatus* (increase, $p = 0.05$) and axillary sea bream, *Pagellus acarne* (decrease, $p = 0.009$). Like in East Corsica, the only community indicator with a significant trend was the size spectrum slope, which increased (Figure 3c).

North Sea and English Channel shelf communities

For the Eastern Channel and Southern North Sea communities, there were signs of worsening impacts of fishing on several populations, whereas community-level indicators remained stable (Tables 6 and 7, Figure 3a). In both communities the size spectrum was curved and bumpy, so linear regressions were not fitted as the interpretation of variations in slopes and intercepts would be senseless.

One population decreased in abundance in the Eastern Channel, black bream (*Spondyliosoma cantharus*, $p = 0.04$), which had a strong year class in 1997, the first year of the data series (Forest, 2001). Two populations were found to decrease in average length (Table 7), herring (*Clupea harengus*), probably due to the strong 2001 and 2002 year classes (ICES, 2003), and whiting (*Merlangius merlangus*). Simply applying the assessment rule of Table 4b, the populations are concluded to be overall deteriorating with $p = 0.039$. However, if the trends for black bream and herring are not attributed to fishing, the number of deteriorating populations is one only, and the conclusion that overall the populations are not deteriorating cannot be rejected ($p = 0.37$).

In the Southern North Sea, two populations were found to decrease in abundance, poor cod (*Trisopterus minutus*, $p = 0.002$) and thornback ray ($p = 0.04$), whereas the grey gurnard population was increasing ($p = 0.01$). Two populations had decreasing average length, plaice ($p = 0.001$) and lemon sole (*Microstomus kitt*, $p = 0.02$).

Atlantic shelf communities

In the Celtic Sea and Bay of Biscay, many populations increased in abundance and many decreased in average length (Table 8). Both observations applied to imperial scaldfish,

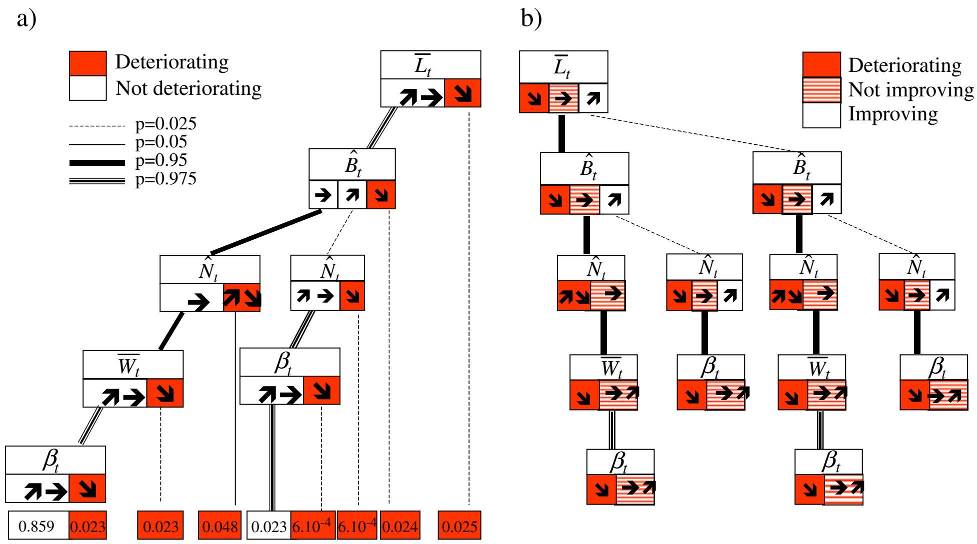

Figure 2. Diagnostic trees for the identification of desirable/undesirable combinations of trends in community indicators when initial state (a) bears no strong fishing impact; (b) is strongly impacted by fishing. One trend test is examined at each knot, and a branch is selected depending on the test result. Cells without a further branch are end-points. Cells are shaded if one of the potential mechanisms for trends is increasing fishing impact, hatched if there is suspicion of stationary fishing impacts. For case (a) the probability of ending in each end-point cell under the null hypothesis of a stable neutral community is given at the bottom. For case (b) they can be calculated as the product of probabilities associated with each branch of the path leading up to the cell under consideration.

Arnoglossus imperialis, in the Celtic Sea; wedge sole, cuckoo ray, *Leucoraja naevus*, and smallspotted catshark, *Scyliorhinus canicula*, in the Bay of Biscay, a possible explanation being an increase in recruitment. Many species changed in either length or abundance, including pelagic, benthic, and demersal species. Some were commercial target species like monkfish (*Lophius piscatorius* and *L. budegassa*) and dab, some were typical discarded bycatch like boarfish (*Capros aper*), and others had a low commercial value. Four species had similar positive trends in both ecosystems (imperial scaldfish, spotted dragonet, *Callionymus maculatus*, conger eel, *Conger conger*, and thickback sole, *Microchirus variegatus*).

The community indicators were not completely consistent with the population indicators (Table 7 and Figure 3d,e). In the Bay of Biscay, 40% of the populations examined increased in abundance, but total abundance did not increase significantly, whereas community biomass did increase significantly (Figure 4), along with the intercept of the size spectrum (suggesting there were more animals of any size in the community). The inconsistency between the significant exponential increase in the abundance of 20 populations and stationary community abundance is due to the dominant species in the community not increasing, and the linear model not detecting the increase in community abundance, because of an outlier in 1994 and the S-shape of the time-series (Figure 4). We conclude that fishing impacts were not reduced in the Bay of Biscay, although the hypothesis of a stable community is rejected (Figure 3d, Table 6).

In the Celtic Sea, there were as many decreases in length as increases in abundance, and this resulted in a decrease in the average weight in the community, although total abundance did not decrease nor total abundance increase, again because of the low power of the linear model for this short time-series (Figures 3e and 4). In addition, the size spectrum intercept decreased, suggesting a decrease in numbers at all sizes. Overall, there is a strong signal of an increasing impact of fishing in the Celtic Sea, on both fish populations and the community (Table 6).

Discussion

Fish communities

The overall picture of exploited fish communities around France is that impacts of fishing are steady or increasing in eight of the nine communities examined. Only two communities, the shelf east of Corsica and the Vilaine Estuary, were diagnosed as moderately impacted at the beginning of the assessment. This satisfactory state persisted only off East Corsica, but we detected a deterioration in populations of the Bay of Vilaine. In the other seven ecosystems, already impacted communities were not recovering. It is worth noting that all these communities support, among others, ongoing trawl fisheries, whereas East Corsica supports smaller scale fisheries using more selective and less destructive gears (Abbes, 1991).

Both Mediterranean communities were stationary, but the final diagnostics contrast with the initial states. In the

Table 6. Assessment of initial state and results of the combined trend diagnostics for nine fish communities around France.

Community	Initial year	Initial state	Description and sources	Trends in populations	Trends in community	Overall diagnostic
Seine Estuary	1995	Impacted	Fish habitat loss and low suitability (Riou *et al.*, 2001) Destructive shrimp and flatfish fisheries (Bessineton *et al.*, 1994)	Not improving	Not improving	Not improving
Somme Estuary	1999	Impacted	Destructive shrimp and flatfish fisheries (Bessineton *et al.*, 1994)	Not improving	Not improving	Not improving
Vilaine Estuary	1982	No strong impact	Decline of the shrimp fishery in the early 1980s (Forest, 1988) Moderate decline in diversity of fish species of commercial interest (Désaunay and Guérault, 2003)	Deteriorating ($p = 0.0128$)	Stationary	Deteriorating
East Corsica	1995	No strong impact	Low fishing activity in the 1980s (Lebeau, 1986) and no signal of extension since then (Relini *et al.*, 1999)	Stationary	Stationary	Stationary
Gulf of Lions	1995	Impacted	Severe fishing impacts since the 1970s (Meuriot *et al.*, 1987; C.G.P.M., 1988) Major stocks in a poor state (Aldebert *et al.*, 1993)	Not improving	Not improving	Not improving
Southern North Sea	1990	Impacted	Too high levels of exploitation (ICES, 1991a) North Sea cod at risk of collapse (Cook *et al.*, 1997) Long-term fishing impact on communities (Rijnsdorp *et al.*, 1996; Jennings *et al.*, 2002) High levels of discarding (Stratoudakis *et al.*, 1998)	Deteriorating ($p = 0.0026$)	Not improving	Deteriorating
Eastern Channel	1991	Impacted	Flatfish and gadoid stocks outside safe biological limits (ICES, 1991a), due to high fishing mortality rates and high levels of discarding (Mellon, 1998)	Not improving	Not improving	Not improving

Celtic Sea	1997	Impacted	A majority of stocks outside safe biological limits (ICES, 1998) Long-term fishing impact on community (Pinnegar *et al.*, 2002) High level of discarding (Rochet *et al.*, 2002)	Deteriorating ($p = 0.00016$)	Deteriorating ($p < 0.021$)	Deteriorating
Bay of Biscay	1987	Impacted	Increasing exploitation level especially on young fish, too small mesh sizes (ICES, 1991a)	Not improving	Not improving	Not improving

Gulf of Lions in the early 1970s, the demersal resources were still considered under-exploited (Bonnet, 1973). First diagnoses of overexploitation there occurred after the rapid development of a bottom trawling fleet in the mid-1970s (Meuriot *et al.*, 1987). By the beginning of the MEDITS survey, severe impacts had already accumulated (Table 6), and the present results show that they have not lightened. By contrast, the shelf east of Corsica is narrow (9 km wide, surface area 1432 km^2, compared with 74 km and 11 262 km^2 for the Gulf of Lions) and has a rough bottom, and the island market has a limited demand for demersal fish. As a result, a trawler fleet did not develop and fishing activities in this area remained at a small scale, with potential impacts limited to the target species (snappers, spiny lobster).

Changes in both the direction of increasing fishing impacts and the converse were detected for the three Atlantic communities examined. Parts of these changes might be ascribed to the increase in ocean temperature over the last 30 years (Koutsikopoulos *et al.*, 1998; Planque *et al.*, 2003). There is indeed evidence of a strong influence of hydro-climatic conditions on fish communities in this region, e.g. winter–spring freshwater supply determines spatial distribution and abundance of certain populations (Le Pape *et al.*, 2003). The hypothesis of climate-induced changes in species abundance was thoroughly examined for the Vilaine Estuary and the Bay of Biscay, and could not be rejected (Désaunay *et al.*, in press). Another related consequence of the warming trend is an increase in the number and abundance of tropical species as well as species that are at their northern distributional limit in the Bay of Biscay (Quéro *et al.*, 1998; Poulard and Blanchard, in press), together with a decline in boreal/temperate species (Poulard and Blanchard, in press). In addition, we also detected typical signs of increasing fishing impacts in the Celtic Sea, which were consistent with previous studies showing that in the Celtic Sea undersize fish are caught, most of which are discarded (Rochet *et al.*, 2002; Trenkel and Rochet, 2003), and that the long-term fishing-induced decline in trophic level of the community is ongoing (Pinnegar *et al.*, 2002). According to commercial stock assessments using age-structured catch and effort models, most stocks in the Celtic Sea are overexploited (ICES, 2002).

Both coastal and northern shelf-sea communities bear strong fishing impacts, and deteriorated over the study periods. The North Sea is well-known as a highly impacted ecosystem (see references in Table 6) and the situation is not improving (ICES, 2003). This impact extends to the adjacent English Channel, partly because some fleets and stocks occupy the two areas. Species such as whiting and plaice, which have been diagnosed as overexploited in standard stock assessments (ICES, 2003), were also found here to be impacted by fishing. We also concluded that species such as poor cod, black

Table 7. Combined trends in population indicators. In each cell is the number of populations with the corresponding combination of trends. For shading and hatching coding, see Tables 4 and 5.

Seine Estuary	$\ln(\hat{N}_{i,t})$		
$\bar{L}_{i,t}$	↗	→	↘
↗			
→		9	
↘			

Somme Estuary	$\ln(\hat{N}_{i,t})$		
$\bar{L}_{i,t}$	↗	→	↘
↗			
→		9	
↘			

Vilaine Estuary	$\ln(\hat{N}_{i,t})$		
$\bar{L}_{i,t}$	↗	→	↘
↗			
→	2	6	3
↘			

East Corsica	$\ln(\hat{N}_{i,t})$		
$\bar{L}_{i,t}$	↗	→	↘
↗		1	
→		19	
↘		2	

Gulf of Lions	$\ln(\hat{N}_{i,t})$		
$\bar{L}_{i,t}$	↗	→	↘
↗		1	
→		19	1
↘		1	

Southern North Sea	$\ln(\hat{N}_{i,t})$		
$\bar{L}_{i,t}$	↗	→	↘
↗			
→	1	8	2
↘		2	

Eastern Channel	$\ln(\hat{N}_{i,t})$		
$\bar{L}_{i,t}$	↗	→	↘
↗			
→		14	1
↘		2	

Celtic Sea	$\ln(\hat{N}_{i,t})$		
$\bar{L}_{i,t}$	↗	→	↘
↗		1	
→	8	24	1
↘	1	8	

Bay of Biscay	$\ln(\hat{N}_{i,t})$		
$\bar{L}_{i,t}$	↗	→	↘
↗		3	
→	17	24	1
↘	3	3	

bream, and thornback ray, which are not formally assessed and for which little or no information other than survey data are available (Forest, 2001) were impacted by fishing. In addition, the English Channel and adjacent coastal communities suffer from pollution and other human impacts (Rybarczyk, 1993; Desprez, 2000), especially Seine Bay (Abarnou *et al.*, 2000). Impacted habitats might impair sensitive processes in the life cycle and limit the resilience of communities to fishing impacts.

One species, thornback ray, changed significantly in several of the communities examined, either in abundance (decreasing in the Gulf of Lions and North Sea) or in average length (decreasing in the Bay of Biscay, increasing off East Corsica). This is not surprising because this species has long been known to be sensitive to fishing owing to its particular life history characteristics (Walker and Hislop, 1998). In the Gulf of Lions a marked decline in most commercial elasmobranch species has been noted since the 1980s (Aldebert, 1997).

The combined indicator trends method

Our proposal for combining trends provides a picture of changes in the community as a whole, not just commercially targeted species. As such, it is complementary to the formal assessment of target stocks, and might give early signals of ongoing changes on different scales. The method has the advantage of clearly separating the estimation and assessment steps. For the estimation method to give a reliable picture of community dynamics, it depends on (i) an adequate interpretation of combined trends in a set of relevant indicators, (ii) the availability of sufficient data, (iii) an appropriate method to detect trends. Further, the final outcome of the assessment will depend on (iv) an adequate assessment of the initial state of the communities, and (v) the decisions made about acceptable and undesirable combinations of trends.

The key step of our approach is to provide a conceptual framework for interpreting combined trends, aiming at making the most of the information enclosed in the indicators. The framework developed here for a small number of indicators should be refined. As the number of indicators increases, the number of potential combinations does so in a factorial manner, but the number of potential interpretations in each cell decreases. The use of sequential methods with end-points as soon as a firm interpretation can be reached should limit this problem. In this respect, the approach works like an identification key: starting from broad characteristics and refining until a final identification can be made. We face the challenge of developing a taxonomy of

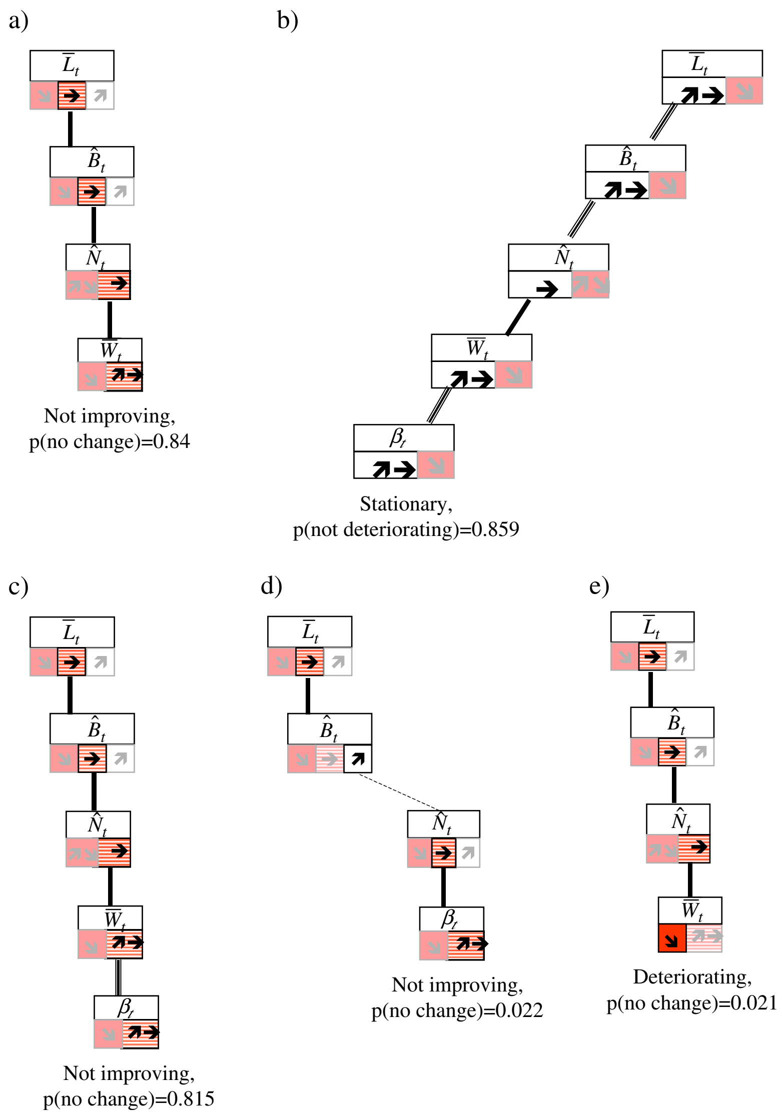

Figure 3. Diagnostic trees of combined trends in community indicators. (a) Somme and Seine Estuaries, Southern North Sea, and Eastern Channel; (b) Bay of Vilaine and East Corsica; (c) Gulf of Lions; (d) Bay of Biscay; (e) Celtic Sea. Shading and hatching coding as in Figure 2. Only the followed assessment paths are shown.

fishing impacts on populations and communities, and determining appropriate sequences of indicators. Which indicators are to be examined first depends on their relevance to the management objectives, their measurability (availability of data and precision of indicator estimates), and the time scale considered: the processes listed in Table 3a will respond to fishing and/or the environment on different time frames. Unlikely combinations (such as decreasing average weight and increasing average length) should trigger further investigations and a check of the field and estimation methods. Thinking both backwards (starting from potential combinations and listing potential mechanisms as done here) and forwards (starting from factors and predicting their joint effects on indicators) is necessary to increase the chances of establishing a complete interpretation table. Population and community models would be useful for this step.

This first attempt to combine trends was limited by the restricted set of indicators used. Complementary indicators at the population level would help distinguish between the various potential explanations of each combination of trends in log abundance and mean length (Table 3a). For example, as exemplified by Channel herring, further descriptors of length distributions, e.g. a low and a high percentile, would help to determine whether decreasing average length is due to more small fish or less large fish or both. Similarly,

Table 8. Slope estimates of trends in population indicators for the Celtic Sea and the Bay of Biscay. *: $0.01 < p \leq 0.05$, **: $0.001 < p \leq 0.01$, ***: $p \leq 0.001$.

Species	Bay of Biscay, 1987–2002		Celtic Sea, 1997–2002	
	Trend in $\ln(\hat{N}_{i,t})$ (y^{-1})	Trend in $\overline{L}_{i,t}$ (cm y^{-1})	Trend in $\ln(\hat{N}_{i,t})$ (y^{-1})	Trend in $\overline{L}_{i,t}$ (cm y^{-1})
Argentina silus		0.48*	−0.54*	
Argentina sphyrena				−0.52*
Arnoglossus imperialis	0.23***		0.38*	−0.37*
Arnoglossus laterna	0.12*			
Buglossidium luteum	0.24**			
Callionymus lyra			0.23**	
Callionymus maculatus	0.18**		0.26*	
Capros aper	0.17*			
Cepola macrophthalma	0.15***			
Chelidonichthys cuculus				−0.38*
Clupea harengus				−0.52**
Conger conger	0.17***		0.20**	
Dicentrarchus labrax	0.10*			
Dicologoglossa cuneata	0.17***	−0.26*		
Echiichthys vipera	0.20***			
Enchelyopus cimbrius	0.18**			
Gadiculus argenteus	−0.23*			
Galeus melastomus	0.19***			
Helicolenus dactylopterus dactylopterus		−0.31*		
Lepidorhombus whiffiagonis				−0.46**
Leucoraja naevus	0.09*	−0.22*		−1.60*
Limanda limanda			0.37*	
Liza ramada		0.27**		
Lophius budegassa			0.14*	
Lophius piscatorius			0.23*	
Melanogrammus aeglefinus			0.56*	
Microchirus variegatus	0.17***		0.17*	
Microstomus kitt				−0.30*
Molva molva				−11.37*
Pleuronectes platessa				−1.14**
Raja clavata		−0.89**		
Sardina pilchardus	0.10*			
Scomber scombrus	0.12*			
Scyliorhinus canicula	0.10**	−0.34*		
Solea solea	0.12***			
Spondyliosoma cantharus	0.14**			
Spratus spratus				0.38*
Trachinus draco		−0.52***		
Zeus faber	0.13*			

properly combined indicators of individual growth rate, mortality, recruitment, and stock reproductive capacity would allow us more firmly to ascribe detected trends to fishing or to other causes. Obviously, fishing pressure and environmental indicators such as water temperature or food abundance could bring useful additional information. The idea is to eliminate potential causes until the most plausible mechanisms of trends are identified.

Because of their non-specificity, indicators cannot be examined independently for their response to fishing, as done in many previous studies. The question is not whether an indicator is relevant for assessing fishing impacts and the efficiency of management actions, but whether a suite of indicators is. We found the indicators selected here to be generally consistent, except the size spectrum descriptors. In some instances we could not estimate the size spectrum slope because of obvious non-linearity. In other cases, trends were found in the slope of the size spectrum, whereas no other size-based indicator was changing. This is probably due to the size spectrum being estimated only from the

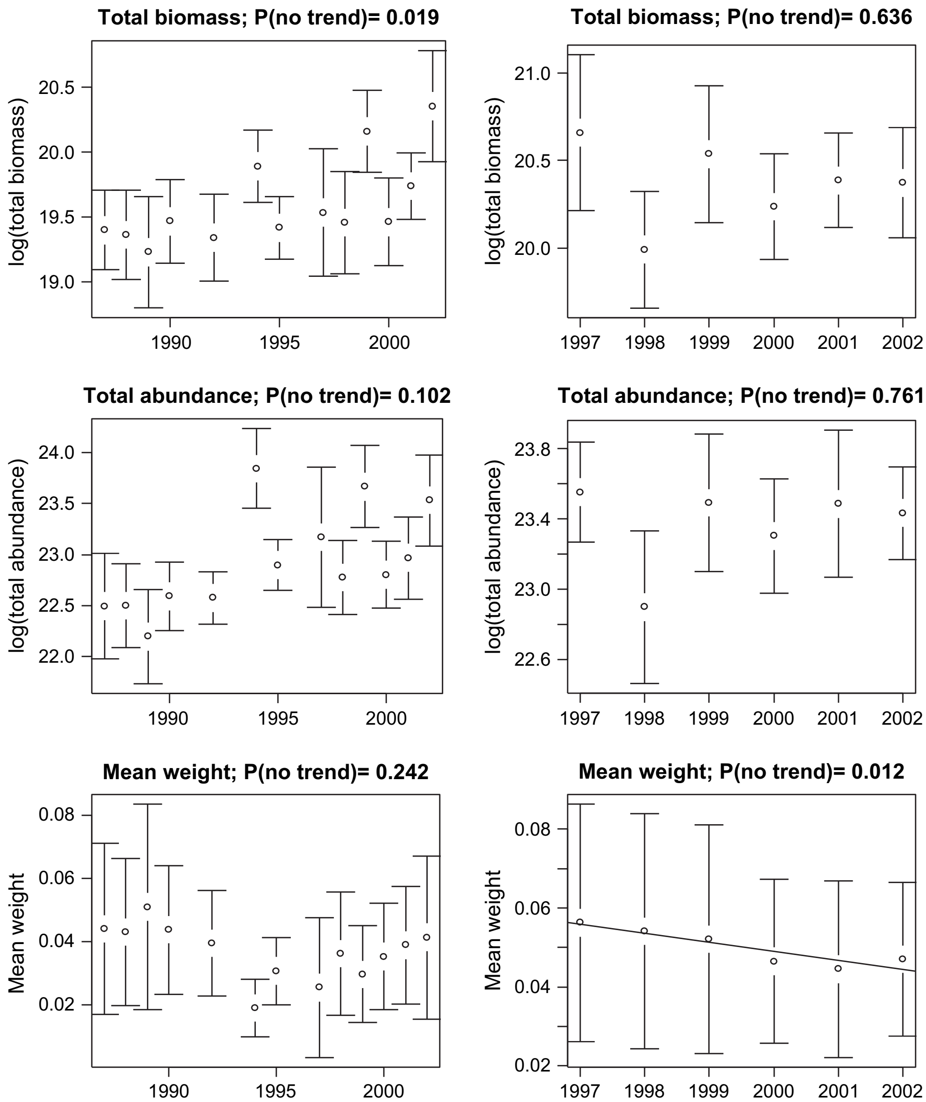

Figure 4. Community indicator trends in the Bay of Biscay (left) and Celtic Sea (right): total community biomass (kg, top); total community abundance (middle); average weight in the community (kg, bottom). Error bars: 95% confidence intervals. Line: fitted linear trend, when significant.

fully recruited size classes, contrary to the other indicators which include all fish caught. In addition, the generally accepted expectation that fishing should decrease size spectra slopes might be due to an oversimplification of the processes governing size spectra dynamics (Benoît and Rochet, 2004). We conclude that size spectra slopes might not be very useful as short- to medium-term indicators of ecosystem dynamics, because their meaning is not obvious. Descriptors other than slope might be more appropriate. This study also shows that the appropriateness of indicators depends on the characteristics of the community being assessed and the expected effects of fishing on it. For example, in the coastal nurseries there is a continuous input of small fish; fishing should not affect the size composition of the community as much as abundances and species composition. This is illustrated by the six populations that changed in average length in the Eastern Channel and the Bay of Biscay, whereas they did not in the adjacent estuaries. Thus, the size-based indicators do not convey much information about nurseries.

One very important point for the implementation of the indicator trends approach is to have at hand a long enough time-series of survey data with a consistent design. From the examples shown it is obvious that 4 years is not enough. On the other hand, the approach is meant for detecting current and not long-term changes. However, our capability to actually detect trends depends on the power of the trend tests, which in turn depends on the variability of the indicator estimates. Unfortunately, the power of such tests is generally low. Nicholson and Jennings (2004) found that around 14−16 years of IBTS (International Bottom Trawl Survey) would be required to detect changes in mean length or weight, given the rate at which these were estimated to change in the North Sea. Hence, 10−15 years in a data series seems adequate, although not easy to achieve with a constant survey design. For the EVHOE survey, the scientific vessel changed in 1997. However, inter-calibration experiments showed that the difference between vessels was lower than the uncertainty attributable to spatial heterogeneity and natural fluctuations (Pelletier, 1998). The fishing gear and most other characteristics of the survey design were kept constant, so the whole series was used for the Bay of Biscay. At the same time, in the Celtic Sea the spatial allocation of hauls was changed and this led us to remove the earlier part of the series. All of the indicators examined here, the abundance indices (at least those of the most abundant species which are estimated with a sufficient precision), as well as all the size-based indicators, are sensitive to changes in survey method (gear, design, etc.) (Trenkel *et al.*, 2004). As soon as there are changes in the survey method, it becomes difficult to determine whether the trends in the indicators are ascribable to the changes in the method or to other impacts. Hence, we recommend that survey methods be kept as consistent as possible.

A limitation of the approach is the power of the tests to detect trends, which is closely linked to the issue of the length of the time-series. Linear regressions were used to detect trends in indicators. Obviously, this is generally not the best model to describe temporal variations (e.g. Figure 4). However, the purpose of the trends method is not to provide a good fit to the data series, but to decide whether there has been a change in an undesirable direction. Testing for linear trends will be powerful in most cases. The combination of test results provides a partial remedy to the power problem. As illustrated in Figure 2a, the probability of a false alarm (concluding that a community deteriorated whereas it actually did not) after a sequence of five tests is 0.12. The benefit of this high α risk is an increase in power (although the latter is not easy to quantify). This might be adjusted by changing the α risks of each indicator trend test, depending on the level of conservativeness desired in the assessment. A risk-averse approach, i.e. favouring the detection of changes, would use high α risks resulting in a higher power, while a risk-prone approach would lead to a lower power while taking a lower risk of false alarm.

On the diagnostic side, the example of the Mediterranean communities clearly shows that the conclusions are highly dependent on the initial assessment. Whereas both communities show similar trends in indicators, the final diagnoses are dramatically different. Given its determining influence on the final diagnostic, the initial state must be carefully assessed. Here, we performed a rapid appraisal based on published evidence. However, this might not always be possible, e.g. for less well-known ecosystems. More formal approaches could be developed to combine retrospective information from diverse sources. These could include catch and effort data, the results of stock assessments, appraisals of damaging practices like discarding, and even the indicators themselves considered in a different way (e.g. multivariate reference space). Performing this analysis afterwards, one can take advantage of validated and published knowledge. The approach would have to be adapted to local settings, because the availability of information can vary greatly between regions (e.g. Table 6). More importantly, it is imperative for fisheries managers to specify the criteria for deciding whether a community is impacted by fishing, in relation to management objectives. Whether only the exploitable stocks are to be taken into consideration, or, if not, which other species or habitats, or general aspects of ecosystem health, are policy issues, not scientific questions.

Finally, the overall diagnostic is determined by the decisions made while defining the acceptable and undesirable combinations of trends. To exemplify the method, we made arbitrary choices in deciding how many diagnostics (deteriorating/improving) would be relevant to each initial state, and which cells in our tables or trees would deserve these diagnostics. This step typically pertains to the realm of management objectives and their translation into indicators. Therefore, in the real world it is up to managers and/or stakeholders to make these decisions. Undesirable trends could depend on environmental fluctuations that change system productivity and the potential harvest, independently of fishing effort (Caddy and Seijo, 2005). Undesirable trends could also be adjusted to a finer assessment of initial status. For example, because in the Celtic Sea it was known from the outset that small fish would be caught, a further decrease in average length of populations could have been set as "deteriorating" instead of "not improving", even when population abundance increases. Conversely, examining which indicators contributed to the final diagnostic could help suggesting appropriate management measures. For example, in the Celtic Sea, the decrease in average length at both population and community levels would indicate that an increase in mesh sizes and/or measures aiming at reducing discards should be taken together with a reduction in fishing effort.

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