Can simple be useful and reliable? Using ecological indicators to represent and compare the states of marine ecosystems

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Within the IndiSeas WG, the evaluation of exploited marine ecosystems has several steps, from simple binary categorization of ecosystems to a more-complex attempt to rank them and to evaluate their status using decision-tree analyses. With the intention of communicating scientific knowledge to the public and stakeholders, focus is on evaluating and comparing the status of exploited marine ecosystems using a set of six ecological indicators and a simple and transparent graphic representation of ecosystem state (pie charts). A question that arose was whether it was acceptable to compare different types of marine ecosystems using a generic set of indicators. To this end, an attempt is made to provide reference levels to which ecosystems can be objectively compared. Unacceptable thresholds for each indicator are determined based on ecological expertise derived from a questionnaire distributed to a group of scientific experts. Analysis of the questionnaires revealed no significant difference in the thresholds provided for different ecosystem types, suggesting that it was reasonable to compare states directly across different types of ecosystem using the set of indicators selected.

Keywords: comparative approach, indicators, marine ecosystems, reference levels, thresholds.

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> *"Ce qui est simple est toujours faux. Ce qui ne l'est pas est inutilisable"*
> *("What is simple is always false. What is not is not useful")*
> *Paul Valéry, Mauvaises pensées et autres (1942)*

Introduction

The state of the world's fished marine ecosystems is of concern and presents fisheries managers with major challenges, with more than 75% of the world's fisheries being over- or fully exploited (FAO, 2003). The FAO states that fisheries should be planned, developed, and managed "in a manner that addresses the multiple needs and desires of societies, without jeopardizing the options of future generations to benefit from the full range of goods and services provided by marine ecosystems" (FAO, 2003). In that respect, boundaries for impacts from fishing are ecologically meaningful if harvested populations are kept within ecologically viable levels, if biological diversity is maintained, and if impacts on the structure, processes, and functions of the ecosystem are kept at acceptable levels (FAO, 2003). Achievement of these management objectives under an ecosystem approach to fisheries (EAF) relies heavily on the ability of scientists to evaluate and communicate in a clear and easy form the properties and functions of marine ecosystems, the ecosystem effects of fishing, and the effectiveness of management measures to maintain resources in a sustainable form.

With this aim, fisheries scientists have developed ecosystem indicators that allow synthesis of the theory, empirical, and modelled evidence which underlies the understanding of ecosystem

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status and responses to fishing pressure (e.g. Garcia and Staples, 2000; Cury and Christensen, 2005; Fulton *et al*., 2005; Jennings, 2005; Rice and Rochet, 2005). Ecosystem indicators aim to reflect key ecosystem processes and serve as signals that something more basic or complicated is happening than what is actually measured (NRC, 2000). There is now a wealth of ecosystem indicators proposed by the scientific community, including size-based indicators (Shin *et al*., 2005) and trophic- (Cury *et al*., 2005b) or life-history-based indicators (Jennings *et al*., 1999; Greenstreet and Rogers, 2006), and frameworks have been developed for their careful selection and evaluation of their relevance (Rice and Rochet, 2005; Rochet and Rice, 2005; Piet *et al*., 2008). However, there is still much to do regarding the use of multiple indicators to evaluate the status of exploited marine ecosystems. Indicators serve to assess both the status and trends of ecosystems (NRC, 2000), but emphasis has clearly been placed more on assessing their trends (Rochet *et al*., 2005; Trenkel *et al*., 2007) than on providing ways to assess the current state of ecosystems.

One way to help facilitate ecosystem assessment and the implementation of EAF is through comparative ecosystem studies (Shin *et al*., 2010). Comparisons have been undertaken whereby marine ecosystems have been categorized or ranked according to various criteria. For example, compliance of countries to the FAO's Code of Conduct for Responsible Fisheries (ftp://ftp.fao.org/docrep/fao/005/v9878e/v9878e00.pdf) has been compared recently across 53 countries (Pitcher *et al*., 2009). Another comparative example was undertaken by Coll *et al*. (2008a), who compared fished marine ecosystems using an integrated indicator summarizing the probability that an ecosystem was classified as sustainably fished. The manifold possible comparisons or rankings of marine ecosystems are dependent on the objectives of the comparisons, so interpretation of the results should be viewed carefully in the relevant contexts.

Here, a simple visualization tool is proposed to allow the public to evaluate how heavily or lightly impacted an ecosystem is with respect to fishing effects. It is a simple approach that uses a comparative framework of a set of ecological indicators as the basis to evaluate the ecological states of exploited marine ecosystems. The approach emphasizes both tracking the ecosystem effects of fishing and communicating the scientific knowledge beyond the scientific audience. This is achieved by comparing six indicators chosen to represent recent (2003–2005) ecosystem states (Shin *et al*., 2010) from a wide array of ecosystems, with results presented in pie charts. It is of note that across the many sets of criteria that exist (e.g. Rapport, 1992; Jackson *et al*., 2000; Tegler *et al*., 2001; Rice and Rochet, 2005), all frameworks include the words easily communicable as a desirable feature of any proposed indicator. This is particularly important given that reviews of the strengths and weaknesses of adaptive management (Elzinga *et al*., 1998; Lee, 1999) point to communication failure as one of its greatest potential flaws.

For well-founded comparisons to be made, the generic set of indicators needs to capture the same information across systems and to be similarly interpreted across all ecosystems. In addition, it is important to check whether indicator values can be compared directly across ecosystems, i.e. whether the scale used for representing the indicators can be considered to be the same. This issue can be resolved by determining reference levels for the six indicators studied and by verifying that those reference levels are consistent across ecosystems. Further, the definition of thresholds for ecosystem indicators will allow the identification of ecosystems most likely to be in an undesirable state. To date, however, very few reference levels have been defined for ecosystem indicators, either for characterizing unfished situations, limits to be avoided, or optimal targets for management (Jennings and Dulvy, 2005; Link, 2005). Therefore, to provide a first attempt at defining ecological reference levels and to establish whether there are any differences in reference levels between different ecosystem types that might compromise the utility of cross-system comparisons, we conducted an expert survey of independent fisheries scientists and ecologists.

Material and methods

Selection of indicators

We evaluated the state of 19 exploited marine ecosystems by a suite of ecological indicators (Table 1). The rationale for the selection of

Table 1. Indicator estimates for the period 2003–2005.

Ecosystem	Mean length (cm)	Trophic level of landings	Proportion of under- and moderately exploited species	Proportion of predatory fish	Mean lifespan (years)	$1/CV$ biomass
North-central Adriatic Sea	10.95	3.28	0.19	0.05	5.20	3.09
Central Baltic Sea	22.32	2.13	0	0.05	8.39	5.53
Barents Sea	19.07	3.56	0.13	0.41	20.12	3.73
Bay of Biscay	16.06	3.52	0	0.02		2.11
Southern Benguela	27.10	3.47	0.34	0.23	11.62	2.92
Bering Sea, Aleutian Islands	34.74	3.72	0.21	0.41	32.79	16.43
Canada west coast		3.80	0.15	0.87	36.53	1.36
Southern Catalan Sea	13.70	3.17	0.18	0.32	7.98	4.91
Guinean EEZ	20.14	3.40	0.07	0.78	14.45	2.52
Northern Humboldt		3.34	0.25	0.07	3.56	2.42
Southern Humboldt	24.57	2.76	0.34	0.03	10.04	6.07
Irish Sea	22.62	3.42	0.20	0.92	16.35	1.48
Mauritanian EEZ	26.18	2.80		0.59	12.27	2.88
Morocco (Sahara coastal)		2.99	0.44	0.13	14.58	2.73
North Sea	24.49	3.60	0.20	0.54	5.99	4.24
Portuguese EEZ	16.23	3.28	0.42	0.12	21.85	1.65
Eastern Scotian Shelf	22.15	3.18	0.26	0.71	23.74	6.25
Senegalese EEZ	24.93	3.21	0.07	0.52	11.71	4.52
Northeast United States	15.30	4.01	0.69	0.93	28.94	7.72

those indicators is detailed by Shin *et al.* (2010). In addition to the (emerging) classical selection criteria that encompass ecological meaning, measurability, sensitivity, and public awareness of the indicators (Jennings, 2005; Rice and Rochet, 2005), several other criteria have played an important role. For balanced ecosystem evaluation and graphic representation, there had to be a balance in the types of ecological indicator used (e.g. size-based, trophic, or life-history indicators) and in the ecological features of an ecosystem that can be prioritized into management objectives (Table 2). Shin *et al.* (2010) distinguished three management objectives, acknowledging that there might be some overlap between them because ecosystem processes do not act in isolation: conservation of biodiversity (CB), maintenance of ecosystem stability and resistance (SR) to perturbations, and maintenance of ecosystem structure and functioning (EF). This constraining framework has resulted in the selection of the following six state indicators (see Shin *et al.*, 2010, for a detailed description): mean length of fish in the community (applied to EF), trophic level (TL) of landings (EF), proportion of predatory fish (CB), proportion of under- and moderately exploited species (CB), mean lifespan (SR), and the inverse biomass coefficient of variation, *CV* (SR).

The suite of six indicators is proposed as a means of evaluating the ecological state of marine ecosystems at a point in time, thus providing a comparative snapshot of ecosystem status. It is not identical to that used for studying trends in indicators in response to fishing pressure (Blanchard *et al.*, 2010), because the indicators were used to compare ecosystem status statically, and hence the constraints were more stringent than for trends. The state indicators need to be directly comparable across ecosystems. Therefore, they had to be either dimensionless (e.g. proportion of improving species) or meaningful in absolute terms with respect to the level of fishing pressure (e.g. TL of the landings). For example, the total biomass of a system is used for analyses of trends only (Blanchard *et al.*, 2010; Shin *et al.*, 2010), because in static terms, it does not necessarily reflect the impact of fishing; further, the total biomass is often only available as a biomass index, so is not comparable across ecosystems because the value of the index is ecosystem- and survey-specific.

The set of indicators was calculated by local experts from the 19 ecosystems represented in the first phase of the IndiSeas WG (Shin and Shannon, 2010; Shin *et al.*, 2010), covering different types of ecosystem and different oceans (upwelling, temperate, high-latitude ecosystems from the Atlantic and Pacific Oceans, and the Mediterranean Sea). Because of the high level of variance in the indicators, it was decided that it would be more appropriate to represent the present state of ecosystems by considering the mean values of indicators over a recent period of 3 years (a moving time-window), instead of considering only a point (1-year) estimate. The values were therefore averaged over 3 years during the most recent period common to all the ecosystems, namely 2003–2005 (Table 1).

Polar-area pie charts

To communicate the state of marine ecosystems beyond the scientific sphere, the objective was to choose a meaningful, accurate, and simple representation of the six ecological indicators chosen to represent ecosystem state. Kite and pie diagrams were considered advantageous because they provide simple and multivariate summaries of ecosystems (Garcia and Staples, 2000; Pitcher and Preikshot, 2001; Haedrich *et al.*, 2008). Each branch of the kite or portion of the pie diagram corresponds to a selected

Table 2. Information provided as background and guidance to the scientific experts consulted in the process of defining limit reference levels.

				Temperate ecosystems						Upwelling ecosystems							High-latitude ecosystems	
Ecological criteria for defining ecosystem overfishing	Management objective	Indicator	Calculation	Adriatic Sea NC 1975–2006	Baltic Sea 1974–2005	Catalan Sea S 1978–2005	Irish Sea 1988–2005	North Sea 1983–2006	Scotian Shelf E 1970–2006	Benguela S 1986–2006	Sahara coastal 1998–2005	Guinea 1985–2006	Humboldt N 1996–2006	Mauritania 1986–2006	Portugal 1981–2006	Senegal 1986–2005	Barents Sea 1984–2005	Bering Sea 1978–2006
Significant modification of ecosystem functioning	EF	Mean length of community (cm)	$\sum_i L_i/N$	10.9–12.7	21.1–27.3	13.5–16.8	22.3–26.5	23.7–29.2	18.4–36.6	24.8–28.5	–	13.5–21.6	6.7–30.1	16.4–38.8	11.5–19.1	21.5–29.5	10.6–23.5	29.3–36.4
		TL of landings	$\sum_S TL_S \cdot Y_S / \sum_S Y_S$	3.17–3.34	2.09–2.53	3.08–3.24	3.36–3.75	3.55–4.05	2.79–3.74	3.43–3.73	2.98–3.16	3.22–3.55	2.4–3.16	2.62–3.51	3.21–3.31	3.13–3.35	3.26–3.61	3.7–3.73
Significant decline in biological diversity	CB	Proportion of under- and moderately exploited species	Number of under- and moderately exploited species)/number of target species	0.19	0.33	0.18	0.2	0.2	0.26	0.34	0.44	0.07	0.27		0.42	0.07	0.13	0.21
		Proportion of predatory fish	*B* predatory fish species/*B* surveyed	0.02–0.13	0.05–0.30	0.19–0.35	0.18–0.99	0.24–0.68	0.48–0.91	0.23–0.69	0.11–0.24	0.59–0.84	0.02–0.07	0.52–0.72	0.02–0.21	0.41–0.72	0.10–0.51	0.26–0.52
Significant decrease in stability and resistance of the ecosystem to perturbations	SR	Mean lifespan (years)	$\sum_S (age_{MAX,S} \cdot B_S) / \sum_S B_S$	4.8–5.6	8.2–11.1	7.3–8.7	11.2–20.5	3.6–7.8	18.1–32.5	11.6–19.5	12.9–16.1	9.3–15.5	3.6–7.6	10–13.8	18.9–26.2	9.4–19.6	6.9–22.8	31.7–34.1
		1/*CV* of total biomass	Mean (total *B* for past 10 years) / s.d.(total *B* for past 10 years)	3.09	5.53	4.91	1.48	4.24	6.25	2.92	2.73	2.52	2.42	2.88	1.65	4.52	3.73	16.43

Minimum and maximum indicator values observed are provided for different time-series from different types of ecosystem. EF, ecosystem structure and functioning; CB, conservation of biodiversity; SR, ecosystem stability and resistance to perturbation; *B*, biomass; *N*, abundance; *Y*, landings; *L*, length; *i*, individual; *s*, species.

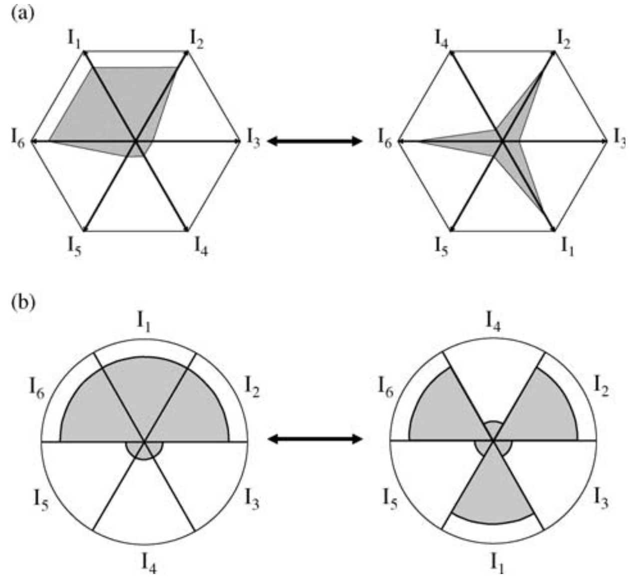

Figure 1. (a) Two kite diagrams and (b) two polar-area pie charts representing the same ecosystem with the same indicator ($I_1 - I_6$) axes. For each type of diagram, the only difference between the left and right diagrams is the order in which the axes are plotted. The total area of the shaded zone is different in the two kite diagrams (a), so depending on the order in which the indicators are plotted.

indicator. As the scientific community is still far from able to determine reference values for ecosystem indicators (though see below), the axis of each indicator displayed is bounded by the minimum (centre of the diagram) and maximum (outer boundary) values observed for respective indicators in each of the set of ecosystems considered. To allow the centre of the graph to be associated with "worse" and the boundaries with "better" state, we transformed some of the indicators. For example, the proportion of overexploited species has been changed to the proportion of low to moderately exploited species. Similarly, the *CV* of biomass was changed to the inverse *CV* of biomass. The boundaries are not intended to represent optimum or target values, but merely to scale the indicators for graphic representation. The crucial point is to adopt the same relative boundaries across all ecosystems to facilitate comparative analyses. The approach underscores the importance of using an inclusive set of case-study ecosystems to obtain meaningful minimum and maximum values for each indicator, reflecting low-to-high levels of fishing pressure.

Although kite diagrams are visually appealing, they can be misleading unless interpreted carefully. In these, the order in which the axes are plotted, for example, can modify our perception of the state of an ecosystem. The total surface of the kite area delimited by the values of the indicators differs depending approximately on the axes (Figure 1a; Rice and Rochet, 2005), because each triangular area between two branches depends on two indicators rather than one. With a pie chart, however, the area of the pies does not change with the order of the indicators because they are independent of each other (Figure 1b). For this reason, pie charts were preferred to kite diagrams and are used here. However, what are referred to as pie charts here and in what follows refers in reality to polar-area pie charts or Nightingale rose diagrams (Brasseur, 2005), sometimes simply called pie slices (Andreasen *et al.*, 2001). For the purposes of visual clarity, the actual minimum value for each pie/indicator observed across ecosystems was given a value of 0.1 in the normalized scale of 0–1. This helps to differentiate between situations where an indicator has reached its minimum value from cases where an indicator is missing. In addition, outer boundaries are not shown for pies with missing values.

Correspondence between pie portions and indicator values when using pie diagrams for representing the state of marine ecosystems is only fully meaningful if the response of the indicators to fishing is monotonic, and preferably linear, across the range of observed values. This is something that has not been checked *a priori*, and the form of the response of indicators to fishing pressure is something that is not well-documented, although some empirical (Blanchard *et al.*, 2005; Greenstreet and Rogers, 2006; Metthrata and Link, 2006) and modelling output (Fulton *et al.*, 2005; Travers *et al.*, 2006) provided some early direction. We therefore refer here to theoretical reference directions of indicators in response to fishing pressure (Jennings and Dulvy, 2005) and consider that the border of the pie diagram should correspond to more desirable values.

Frequency distribution plots were used to evaluate the distribution of the actual indicator values across the 19 ecosystems. Such plots help to evaluate whether the pie diagrams, which are bounded by the minimum and maximum values observed across all ecosystems, are useful for comparative purposes. Indicators for which the frequency distributions are even will tend to have indicator values spread regularly along the portion of the pie diagram, facilitating the differentiation of one ecosystem state from another.

Questionnaire survey of indicator reference levels

To obtain a first approximation of plausible reference levels for the six indicators of ecosystem state, we conducted a science expert survey of experienced fisheries scientists and ecologists. Our rationale was that for the science experts to identify theoretical reference levels, based on their accumulated knowledge and expertise. In the survey, the reference levels were considered to be ecological benchmarks against which ecosystem states could be compared. They were meant to allow comparison of ecosystem states using ecological indicators rather than to be used directly in management decision-making. Further, we considered the reference levels to be limit ones, because it is easier to reach agreement on and to define what is to be avoided than to say what is desirable (Cury *et al.*, 2005a), especially when focusing exclusively on ecological criteria.

Science experts with fisheries, ecological, and indicator expertise were identified by IndiSeas WG members and survey questionnaires were e-mailed to them. The experts were provided with a guide to complete the survey, including a table listing the minimum values observed for each indicator for 15 of the ecosystems (Table 2). As the data for the other four ecosystems (Bay of Biscay, Canada west coast, Northeast US, and southern Humboldt) were not available at the time of the survey, indicator values could not be provided to the experts and are not reported in Table 2. They were encouraged to complete questionnaires only for the ecosystems about which they were most knowledgeable. The type of ecosystem (upwelling, temperate, high latitude) and/or the name of the particular ecosystem the expert was referring to had to be recorded on each questionnaire. Each expert was requested to complete the questionnaire independently (independent threshold values required), i.e. to avoid consensus-seeking discussions with other experts, which would have biased the reference levels that emerged from the survey.

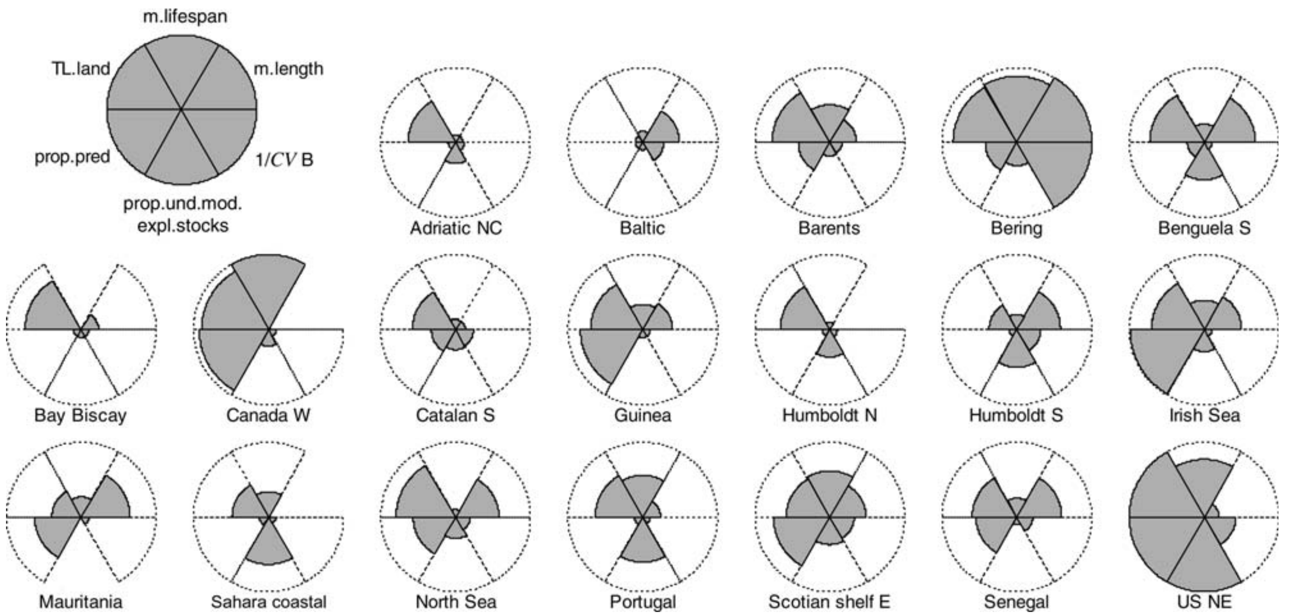

Figure 2. Pie diagrams representing present ecosystem states (2003–2005) using six ecological indicators: mean length, mean lifespan, TL of the landings, proportion of predatory fish, proportion of under- to moderately exploited species, and $1/CV$ biomass. Absence of the external border of a portion of the pie means that the corresponding indicator value was not available.

The questionnaire asked for an unacceptable threshold value from a scientific viewpoint (i.e. a value below which an ecosystem could be considered to be overexploited) to be supplied for each of the six state indicators. Several criteria were proposed for use by the experts in defining ecosystem overfishing in terms of the six ecological indicators (Table 2). These included a significant decline in biological diversity, a significant increase in interannual variation in ecosystem biomass, a significant decline in the resistance of the ecosystem to perturbations, a significant modification of the trophic structure of the ecosystem, and a significant reduction in fish size. In the survey, the reference levels were based on ecological considerations only. Although recognition of the importance of socio-economic concerns for management purposes is acknowledged, explicit handling of these considerations is to be left for other exercises.

To set the unacceptable threshold for each indicator, experts were given two alternatives: either to provide an absolute limit reference level value or to use a relative value. The relative value of the reference level corresponded to the maximum acceptable percentage reduction in the indicator (e.g. for mean length) with respect to the "best" value of the indicator observed in the (type of) ecosystem considered. To guide the experts in their effort, some information was provided (Table 2) about the calculation of the six indicators considered, along with examples of the minimum and maximum values observed in some of the IndiSeas ecosystems (Shin *et al.*, 2010).

Questionnaire analysis

The thresholds proposed by the experts for the six selected indicators from the completed questionnaires were compiled in a data matrix that was then analysed using principal component analysis (PCA). The standard PCA approach is intolerant of missing data because it is based on eigenvalue decomposition of the covariance matrix. Therefore, we used the NIPALS algorithm (non-linear estimation by iterative partial least squares; Wold, 1966), which can handle small quantities of missing data (generally ~5%). Data were normalized before the analysis.

For each type of ecosystem (upwelling, tropical, temperate, and high latitude), a mean position was computed by simple averaging of the scores in corresponding questionnaires and projecting that value onto the PCA graph to detect a potential effect of the type of ecosystem on the answers to the questionnaires. Non-parametric Kolmogorov–Smirnov tests were used to detect potential differences in the thresholds provided for upwelling and temperate ecosystems. Too few values were provided for high-latitude ecosystems and tropical ecosystems to be compared.

Results

Comparative states of marine ecosystems

Representation of ecosystem states using pie diagrams allows a first take at evaluating whether indicator values are balanced within each ecosystem (Figure 2). Of all ecosystems, the eastern Scotian Shelf appears to be best balanced in the values of all six state indicators, which are medium to high compared with other ecosystems. In contrast, there are ecosystems for which all indicator values bar one were at very low levels. For example, in the Bay of Biscay, the mean TL of the catch is high, whereas other indicators were close to the minimum values observed across the 19 ecosystems. The case of Peru and that of the north-central Adriatic Sea and of the southern Catalan Sea are similar. The West coast of Canada is also striking, with three indicators at maximum values, namely mean lifespan, TL of the landings and proportion of predators, whereas two others had minimum values ($1/CV$ biomass, and proportion of under- to moderately exploited stocks).

It is apparent that many ecosystems have unbalanced pie diagrams, with the coexistence of low and high values of indicators (Shannon *et al.*, 2010). This complicates comparison across ecosystems. Therefore, a quick first glance would only allow discrimination into two broad categories: those ecosystems that appear to

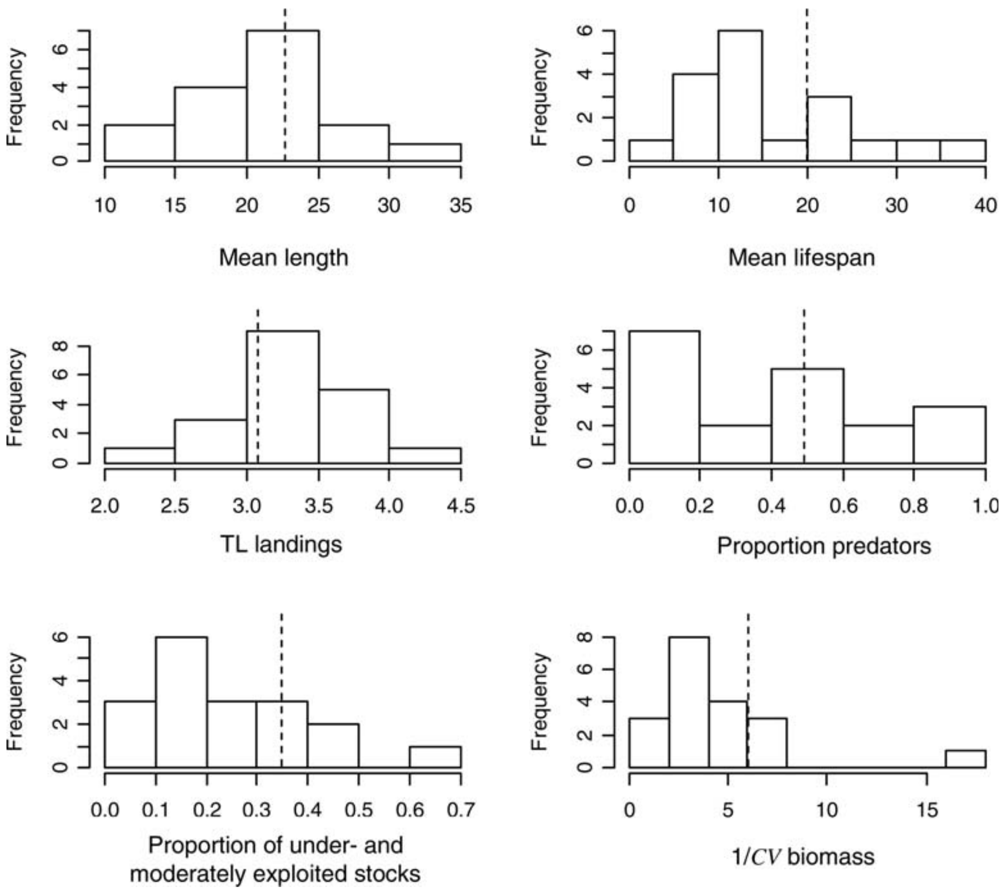

Figure 3. Frequency distribution of indicator values (mean 2003–2005) across the 19 ecosystems studied. Dashed lines represent the mean of the minimum and the maximum values of the indicators.

be currently less influenced by fishing, meaning that at least half of the portions of the pie have high to maximum values and the other half medium values (these are Northeast United States, Bering Sea, eastern Scotian Shelf, and to a less extent the Irish Sea), and those ecosystems that appear currently to be more influenced by fishing, meaning that at least half of the portions are at minimum values and the others at medium values (Peru, north-central Adriatic Sea, southern Catalan Sea, Bay of Biscay, and Baltic Sea).

Distribution of indicator values across ecosystems

Indicators with flat frequency distributions will be more effective for differentiating ecosystems from each other, because the pie values will vary from system to system. An example of such an indicator is the proportion of predators, which has a relatively uniform distribution (Figure 3); other indicators have clear modes in their frequency distributions. For example, the mean length of the community and TL of the landings both have a mode at medium values with a relatively symmetrical distribution, implying that their pie diagrams will display many medium-sized portions and differentiation between the ecosystems will be more marked for either very low or very high values of those indicators. The rest of the indicators (mean lifespan, proportion of under- to moderately exploited species, and $1/CV$ biomass) have asymmetrical frequency distributions that are positively skewed, implying that the majority of ecosystems have small pies for corresponding indicators in the pie diagrams. This seems to indicate that the sample of ecosystems we have is biased towards impacted ecosystems. In addition, for $1/CV$ biomass, the maximum value observed stands out from the rest of the values; the consequence is that it does not discriminate well between the ecosystems, because all ecosystems except the Bering Sea, where this indicator is the maximum observed across all ecosystems, represent small proportions of the pie diagram.

Reference levels

In all, 42 questionnaires were completed by scientific experts and returned for analysis. However, four were not used in subsequent analyses because the type of ecosystem, or its name, was not identified. To avoid issues of bias in the PCA attributable to there being too many missing data per ecosystem, two other questionnaires, in which only one indicator value was provided of the six required, were discarded. Ultimately, therefore, the final data matrix for the PCA consisted of the results from 36 questionnaires and potentially included 216 values, but 19 entries were still missing. The valid respondents included 36 scientific experts from 19 research institutes spread over 17 nations from Europe, Africa, South and North America, Asia, and Australia.

Potential limit reference levels for the six indicators were identified from density plots of the threshold values provided by the questionnaires in the expert survey (Figure 4). Flat distributions indicate no consensus among experts in defining unacceptable thresholds for indicators, at least on an ecological basis. This is

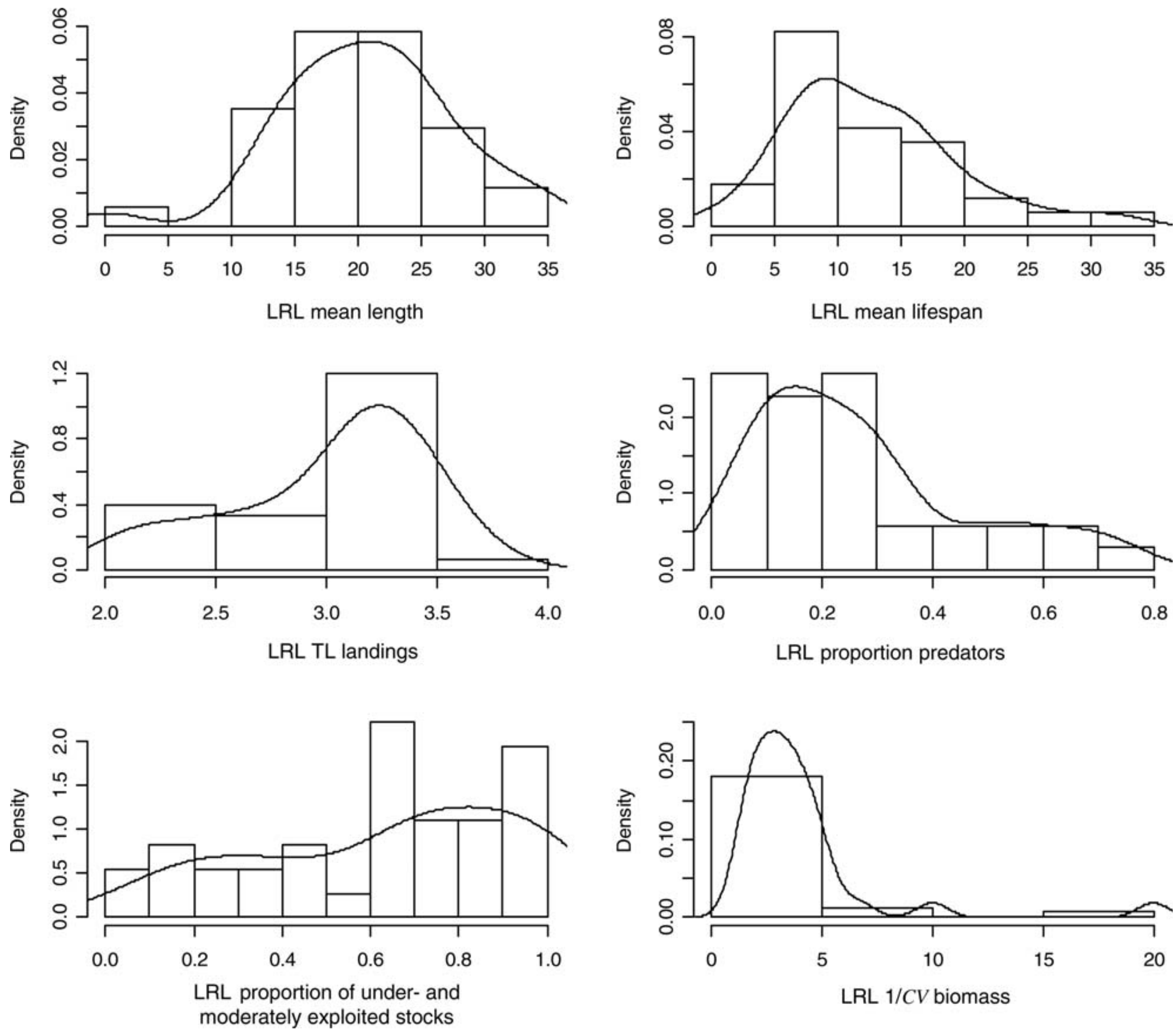

Figure 4. Histogram of density distributions of the threshold values for the six indicators, provided by the surveys of scientific experts ($n = 38$ questionnaires). The lines represent fitted density distributions, and LRL refers to the limit reference level.

the case for the proportion of under- to moderately exploited species, for which the limit thresholds proposed were spread uniformly over the whole range of possible values, although the maximum density was at values from 0.6 to 1. It is of note that all ecosystems except Northeast United States had observed indicator values below this threshold of 0.6, suggesting that they might be highly impacted by fishing.

In contrast, the reference levels for the other indicators had clearer unimodal distributions with, in particular, a leptokurtic distribution for $1/CV$ biomass. The maximum of the fitted densities was at 20.95 cm for mean length, 9.04 years for mean lifespan, 3.24 for TL of landings, 0.15 for the proportion of predators, and 2.82 for $1/CV$ biomass. For seven ecosystems (37%), at least half the indicator values were below these expert thresholds: Bay of Biscay, Sahara coastal (Morocco), the northern Humboldt, the north-central Adriatic Sea, the southern Catalan Sea, the Portuguese EEZ, and the Baltic Sea.

The expert survey was a first attempt to explore whether consensus emerged among marine ecologists for defining limit reference levels for ecosystem indicators, from a strictly ecological perspective. Another aim was to check whether there were differences in these reference levels depending on ecosystem type, the direct implications of this being to determine whether it was reasonable to compare states of multiple ecosystems by simple pies using the same indicators and scales.

To explore whether reference levels were consistent across ecosystem type, the threshold values for the six indicators for temperate and upwelling ecosystems were compared (Figure 5). High-latitude and tropical ecosystems were not considered in this comparison because there was only one questionnaire completed for such ecosystems. No significant differences were detected between threshold values for temperate and upwelling ecosystems at the 5% risk level (Table 3), but a significant difference was detected between upwelling and temperate ecosystems for mean lifespan at a risk level of 10%.

Considering all indicators together in a PCA, similar conclusions were drawn. The first principal component (PC1; 33.25% of the variation explained) is representative of a size

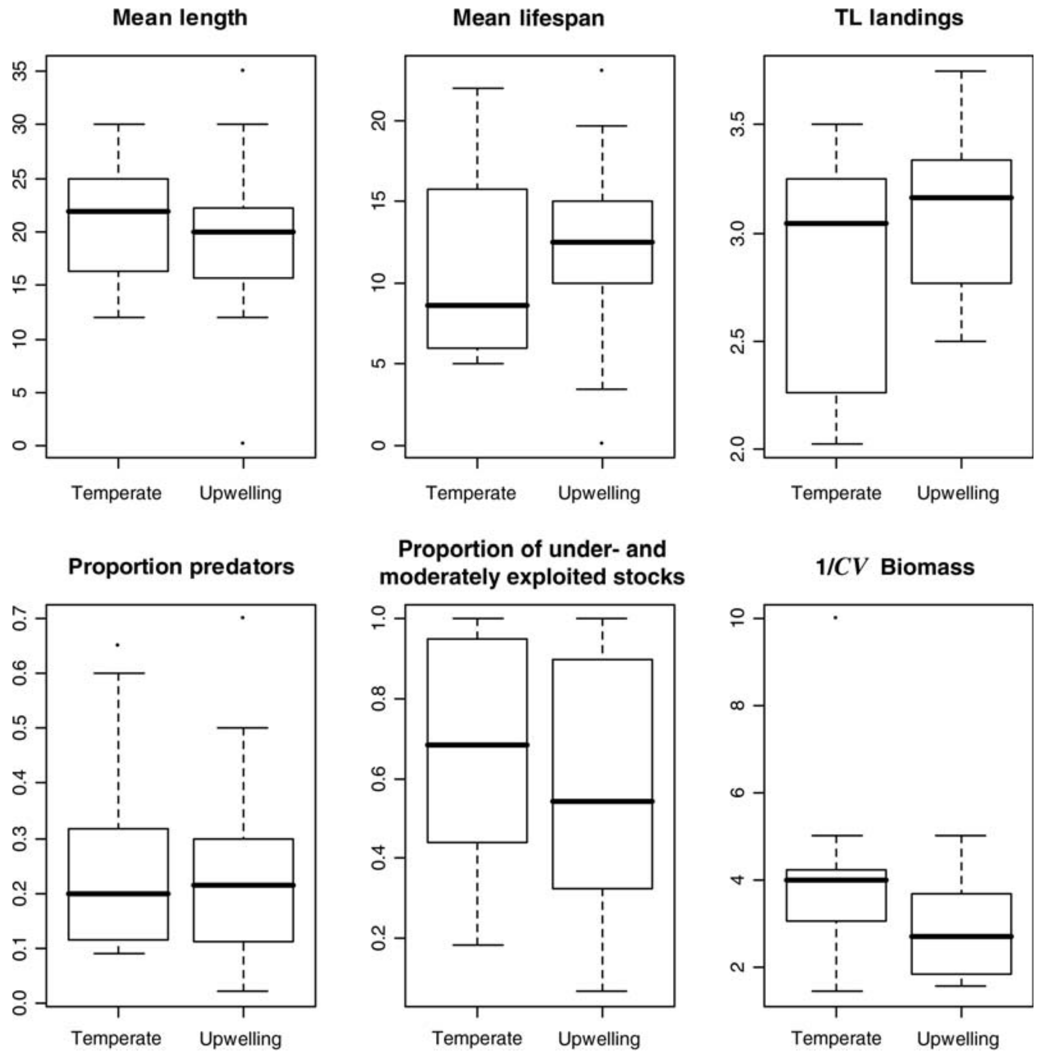

Figure 5. Box and whisker plots of the limit reference values for the six indicators, from surveys of scientific experts (upwelling $n = 17$; temperate $n = 17$). The bold line corresponds to the median, and the box to the interquartile range. The whiskers extend from the box to the most extreme data point, which is no more than 1.5× the interquartile range. Outliers are identified by dots.

Table 3. Non-parametric Kolmogorov–Smirnov test for detecting differences between the threshold values set for temperate and upwelling ecosystems.

Indicator	Kolmogorov–Smirnov test
Mean length	$D = 0.2902$, $p = 0.5133$
TL of the landings	$D = 0.3333$, $p = 0.4313$
Proportion of under- to moderately exploited species	$D = 0.25$, $p = 0.6994$
Proportion of predatory fish	$D = 0.1765$, $p = 0.9651$
Mean lifespan	$D = 0.4664$, $p = 0.07088$
$1/CV$ biomass	$D = 0.3523$, $p = 0.3936$

effect, owing to a correlation between the reference levels set for the six ecological indicators (Figure 6). On the second component (PC2; 22.67% of the variation explained), the proportion of under- to moderately exploited species, mean length, and the proportion of predators were in opposition to mean lifespan and TL of the landings. Projecting the questionnaires on the first two components, the average positions of the two main types of ecosystem documented (temperate and upwelling) hardly differed on the PCA graph (Figure 7). The tropical ecosystem type also fell close to temperate and upwelling ecosystems, suggesting consistency of indicator thresholds across such systems. This is likely related to some homogeneity in the threshold values provided by experts in those three types of ecosystem. Only high-latitude and temperate-to-high-latitude ecosystems stood out on the right side of PC1 owing to globally higher values being attributed to the reference levels in those ecosystems. However, only one ecosystem was documented in each of those types.

Discussion

A simple representation of ecosystem state

The results of this study have illustrated the potential utility and possible applicability of a small set of ecological indicators for comparing the state of exploited marine ecosystems. Simultaneous graphic representation of all six ecological indicators seems to be helpful in communicating scientific information in a

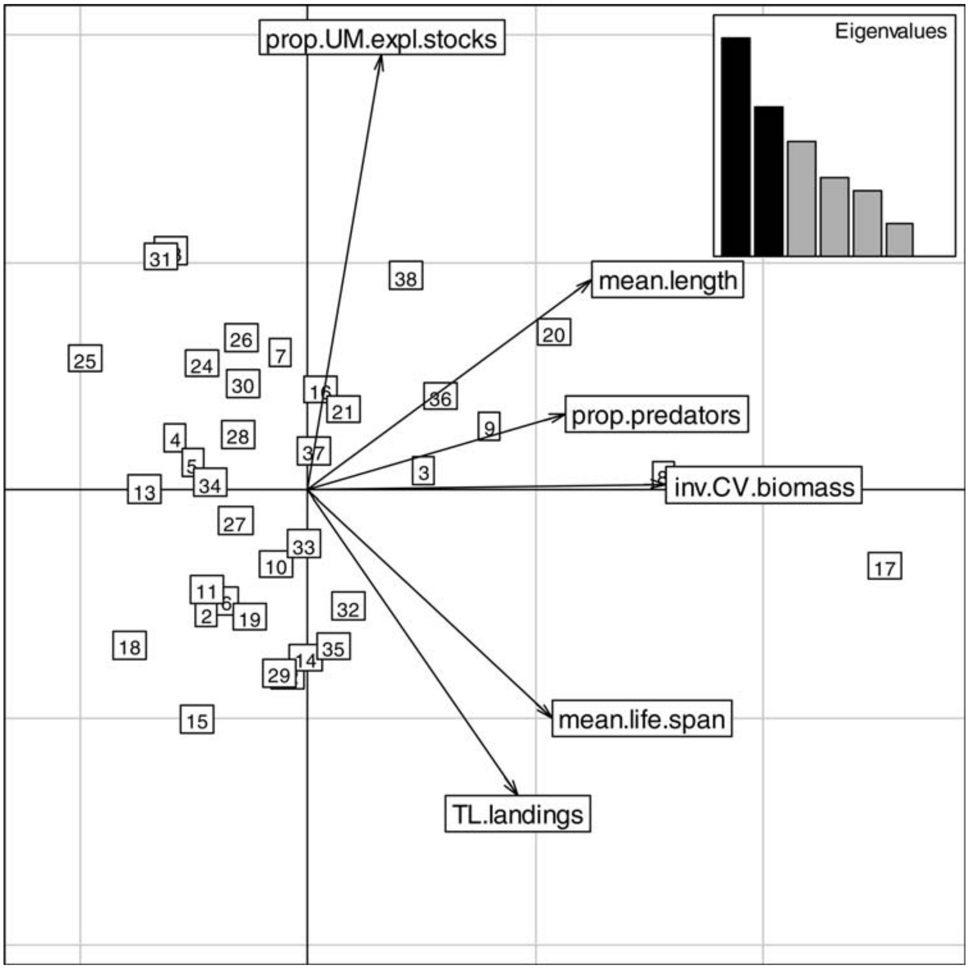

Figure 6. Simultaneous representation of the 36 questionnaires (numbers) and vectors of the six indicators ("prop.UM.expl.stocks", proportion of under- to moderately exploited species; "mean.length", mean length; "prop.predators", proportion of predators; "inv.CV.biomass", $1/CV$ biomass; "mean.life.span", mean lifespan; "TL.landings", trophic level of the landings) onto the first two components of the PCA (horizontally PC1, vertically PC2). The eigenvalues bar plot is given in the top right box; the first two components represented in that box are in black and the others are in grey.

way meaningful to the general public. Pie charts are simple and useful multivariate representations of ecosystem states (Burger and Kelting, 1999; Andreasen *et al.*, 2001), each pie segment representing a particular ecological indicator selected to reflect a facet of EF and which is scrutinized for tracking fishing effects. As for any other form of knowledge transfer, graphics can convey a biased perception of a situation (Gomiero and Giampietro, 2005). This holds true even in our case where pie diagrams were used in a direct and explicit form with few data transformations; multivariate ordination methods such as those used in some other indicator-based approaches (Pitcher and Preikshot, 2001; Link *et al.*, 2002) were not used to synthesize the state of ecosystems, nor were combinations of indicators used to reduce the number of dimensions (Prescott-Allen, 2001).

Some precautions were taken to preclude misleading interpretation of the pie diagrams. First, as multiple indicators were handled and represented, their selection was constrained so as to obtain a final set with a good balance between different ecological features deemed important when evaluating the ecosystem effects of fishing (Shin *et al.*, 2010). Therefore, each portion (i.e. indicator) of the pie diagram was weighted the same in evaluating the state of an ecosystem, making visual inspection of the pies intuitively sensible. Second, as multiple ecosystems were compared using the pie diagrams, it was necessary to adopt the same normalized scale for each indicator across all ecosystems. In the absence of widely accepted reference levels for each indicator, the simplest choice was to adopt observed minimum and maximum values for each indicator across all ecosystems included in the comparison as the boundaries of the indicator axes. This ensured the maximum spread of the ecosystem values along an indicator axis, thus allowing the maximum discrimination between ecosystem values. A consequence of this was that all pies had to be interpreted as relative, rather than absolute, indicators in the framework of a comparative approach. Indeed, the fact that all indicators belonged to exploited systems rendered the maximum and minimum values of indicators decidedly not absolutely good or bad references. Moreover, as more ecosystems are added to this study, the minimum or maximum observed value may change, so changing the relative positions within the pie diagrams.

There are several other caveats to be borne in mind, and a system-specific perspective is always necessary for more detailed interpretation and before general conclusions can be drawn. In

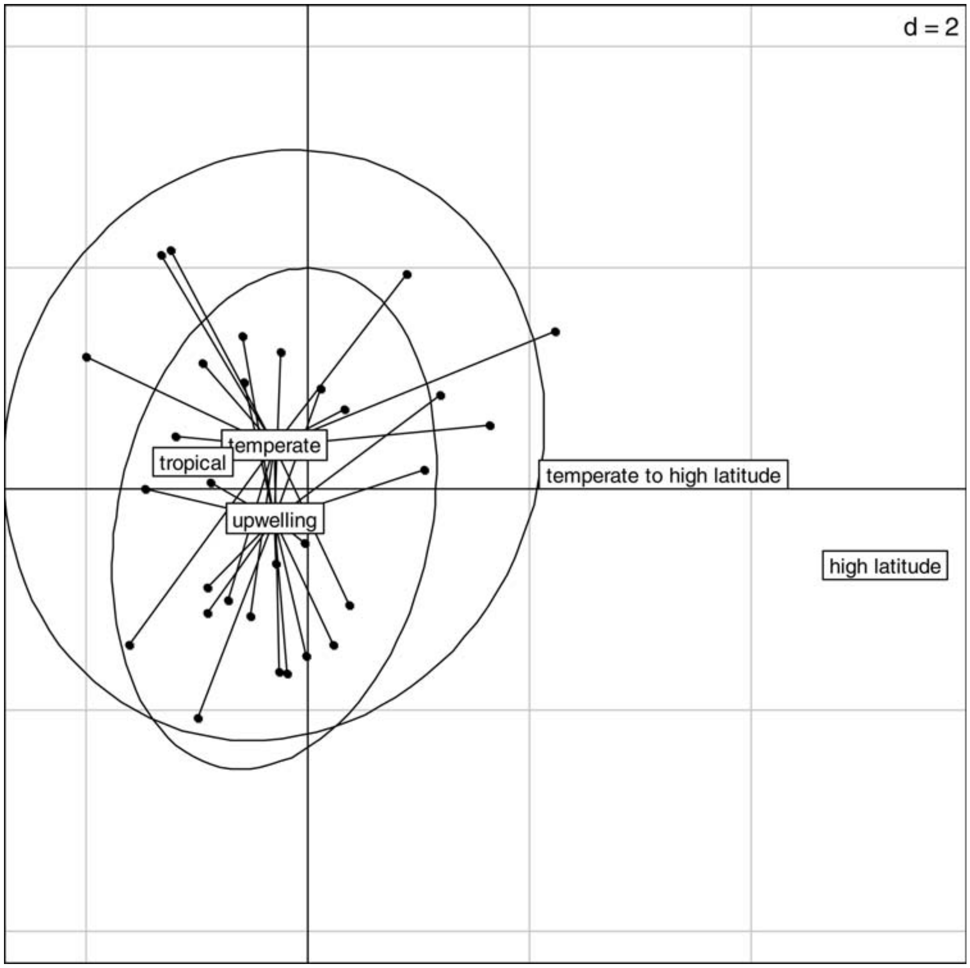

Figure 7. Projection of the average position of ecosystems on the first two components of the PCA. Dots (corresponding to questionnaires) are connected to the average position of their ecosystem type. The ellipses (representing temperate and upwelling ecosystems) are drawn to include 95% of the corresponding questionnaires.

particular, the extent of fishing impacts in each ecosystem, as inferred from the set of indicators under discussion, needs to be carefully considered in the light of non-fishery drivers of ecosystem change, such as environmental or socio-economic factors (see Coll *et al.*, 2010, and Link *et al.*, 2010, for further discussion). There are some caveats linked to how the indicators are represented in the pie charts. Low-to-medium values are not easily discriminated from very low values because the increase in the surface of a portion of a pie increases non-linearly (with an exponent of 2) with the value of the corresponding indicator. This can be problematic in the context of a comparative approach if the distributions of indicator values are right-skewed, meaning that many ecosystem observations would hardly be differentiated on the pie charts. This is the case for the indicator $1/CV$ of total biomass, with almost all values appearing low in the pie diagrams because of one very high value (Figure 3). To a lesser extent, this issue is also associated with the proportion of under- to moderately exploited species and mean lifespan.

An important point that emerges from the results of the present study is that many ecosystems show unbalanced pie diagrams with the coexistence of low and high values of indicators within each (Figure 2), complicating comparison across ecosystems, but also informative. Unbalanced pie diagrams can reveal biases in the calculation of indicators, in the data used, or in the differential response of an ecosystem to fishing. The difficulty inherent in sampling at the scale of a marine ecosystem is obvious whatever the sampled variables are, including those involved in the estimates of indicators presented here. Indeed, even if our indicators are conceptually simple, their estimates remain based on various and often complex data-acquisition processes (Jouffre *et al.*, 2010). The source of the data for the set of indicators calculated here is diverse (e.g. scientific surveys or fisheries data), so the link between the indicators and the data-information system has to be as transparent and explicit as possible (Shin *et al.*, 2010). It is difficult to assess the quality of the data and related information rigorously (e.g. species meta-information and life history traits) and to quantify the ranges of the sampling errors in any way other than with rough expert estimations. The possibility that, in a few cases or on certain indicators, the sampling error can reach the same range as that corresponding to natural variations is not to be ignored. Differences across ecosystems in the way some indicators are estimated should also be recognized, although the same detailed instructions were imposed for calculating indicators in all ecosystems (Shin *et al.*, 2010). For example, the TL of landings reported for the northern and the southern Humboldt may not be directly comparable given the different methodologies used in the two ecosystems to derive the TL of some species (isotope analysis, EwE modelling, and diet analysis; see discussion in Shannon *et al*., 2010). Differences may also be attributed to data availability. Mean length, for example, may not be available in a directly comparable form from different types of survey within a single ecosystem, rendering it possible for that indicator to be estimated from only part of the fish community. This was the case for the mean length derived for the demersal fish assemblage of the southern Benguela. Data availability is also a defining factor in the proportion of under- to moderately exploited species. FAO stock categories were used in addition to FAO assessments for all ecosystems (FAO, 2005), although the information often needed local refinement in each ecosystem considered. For example, the Benguela ecosystem is reported as a single ecosystem in the FAO database, whereas the southern and northern subsystems off South Africa and Namibia, respectively, are not exposed to the same levels of fishing (Cury *et al*., 2005b). In certain ecosystems, such as those off Senegal and Guinea, the stock assessments are less frequent than elsewhere and are available for fewer species (FAO, 2005). To be consistent across ecosystems, we used the FAO reference list of species categories, which did not suit some ecosystems well, because key species were sometimes grouped and less important ones considered separately. Therefore, there is a wide range of caveats to consider when interpreting the results and the data that went into their calculations.

Given those caveats, we believe that unbalanced pie diagrams do reveal that fishing effects (or climate; for that matter, see Link *et al*., 2010; Shannon *et al*., 2010) act on specific parts of an ecosystem (e.g. pelagic vs. demersal assemblages) rather than on the ecosystem as a whole or on certain ecological processes rather than the entire ecosystem functioning (i.e. on only one or a few of the processes captured under the criteria for ecosystem overexploitation in Table 2, rather than on them all). As shown by Blanchard *et al*. (2010), the indicators selected are not redundant, although they are correlated in some way because they are all supposed to vary according to fishing pressure. However, the direction of variation in the indicators does not always conform to theory (Travers *et al*., 2006), because fishing will preferentially target different components of the ecosystem.

Whether these targets are predators or prey, benthic species or pelagic, short- or long-lived species, the indicators under scrutiny will respond differently. For example, emergence of large predatory invertebrates such as jumbo squid (*Dosidicus gigas*) in the Humboldt and California Currents (Field *et al*., 2007; Bograd *et al*., 2008), possibly caused by fishing intensification, would cause indicators such as mean length or TL of the landings to vary in a direction that would be contrary to what would be expected from an increase in fishing pressure. Alternately, mean lifespan would capture this potentially indirect effect of fishing, because the lifespan of jumbo squid does not exceed 2 years (Markaida *et al*., 2005). In the north-central Adriatic Sea, the TL of landings is relatively high, whereas all other indicators are low (Figure 2), reflecting expansion of the fishery on benthic invertebrates (bivalves, mantis shrimp, Norway lobster) as small pelagic fish (sardine, anchovy) declined in abundance. In the Bay of Biscay, all indicator values are low apart from TL of the landings. As the ecosystem includes nurseries for several important stocks (ICES, 2008a), fish are small, and the proportion of predatory fish is low. The relatively high TL of the landings is attributable to the fact that it is calculated from a fixed TL per species, and most TLs are calculated from the adult phase of the species. The multiple processes involved under fishing pressure and the complexity of the responses at an ecosystem-level favour dealing with a panel of complementary indicators rather than a single indicator, whether or not this indicator is integrated. The complementarity of the six indicators selected explains, to some extent at least, the coexistence of low and high values of indicators within the same ecosystem. It could be informative to explore further whether comparing the overall area of the pie slices would provide a useful metric of the relative state of the ecosystem. A further priority could be to develop a decision tree for classifying ecosystems based on the set of six state variables to complement the analysis of Bundy *et al*. (2010) using trend indicators. This would enable the integration of information being captured by the different indicators of ecosystem state to communicate a more solid assessment of fishing impacts across the different ecosystems examined.

An interim approach for defining limit reference levels

This study is one of a few first attempts at proposing limit reference levels for a subset of ecological indicators that may be applied broadly across exploited marine ecosystems for assessing the impacts of fishing. The study should be considered as an exploratory experiment, because it aims to investigate the potential for setting limit reference levels through expert consultation. The survey was intended to explore whether empirical knowledge and the conceptual models of scientific experts about their ecosystem functioning and ecosystem effects of fishing are quantifiable and translatable in terms of indicators and reference levels in a manner consistent across a wide spectrum of ecosystems. As was also argued by Link (2005), the approach aims to make the most of current ecological knowledge to propose first estimates of reference levels; it is a small step that can trigger needed further debate to refine definitions of reference levels and the underlying methods. In the absence of well-defined and well-established reference levels for ecosystem indicators, it has been advocated by others that reference directions be used to provide an assessment of recent trends in indicators and ecosystems (Jennings and Dulvy, 2005; Shin *et al*., 2005). However, even in trends-based analyses (e.g. Rochet *et al*., 2005; Trenkel *et al*., 2007; Bundy *et al*., 2010), there is a need at some stage to characterize the state of an ecosystem, even roughly. In other words, to be able to assess the direction in which an ecosystem is moving, it is necessary to assess from where it is starting. In addition, looking at reference directions alone is not sufficient and can sometimes be misleading (Shannon *et al*., 2009). Indeed, some indicators have a limited range of sensitivities and an ecosystem can be so overexploited that its indicators then do not vary much, e.g. the southern Catalan Sea (Shannon *et al*., 2009), and the direction of change does not always conform with theoretical reference directions, but rather depends on the type of fishing pressure exerted and the type of ecosystem (Travers *et al*., 2006; Coll *et al*., 2010; Shannon *et al*., 2010).

In the present study, the definition of broad reference levels is intended to facilitate comparison of the current states of exploited marine ecosystems. Such a step was perceived as being complementary to the direct comparison of pie diagrams across ecosystems. Therefore, the first goal of the survey was not to provide absolute and precisely defined reference levels that could be used for management, but rather to check whether a consensus could emerge from independent scientific expert judgements on the ranges of indicator values corresponding to the notion of ecosystem overexploitation (Murawski, 2000). Different types of threshold can be used in support of decision-making (Jennings and Dulvy, 2005; ICES, 2008b): target reference levels (reflecting a trade-off between ecological, social, and economic benefits), limit reference levels (associated with serious or irreversible fishing effects on marine ecosystems), and precautionary reference levels (associated with potential harm and taking into account natural variability and uncertainty in the assessment). Reference points can also characterize unfished situations to preclude assessment of the current state of marine ecosystems being complicated by the shifting baseline syndrome (Pauly, 1995). However, such reference points have little value as management targets, provided the levels of resource use are deemed to be acceptable (Jennings and Dulvy, 2005; Shin *et al.*, 2005). To make the comparative approach across ecosystems as meaningful as possible, an option would be to transform each of the six indicators into a ratio over the theoretical value of the indicator in an unexploited state. Unfortunately, this was not a realistic option for the current study because the scientific community is still far from being able to propose effective ways to estimate unfished reference points (Sainsbury and Sumaila, 2003), and ultimately it may not be needed for making management decisions. The few existing studies (e.g. Jennings and Blanchard, 2004; Greenstreet and Rogers, 2006) are not open to generalization for either multiple ecosystem indicators or multiple ecosystems. It was therefore more tractable to attempt to define limit reference levels that could be associated with the notion of ecosystem overexploitation (which refers to ecological processes, not to the socio-economic considerations that usually play a part in defining management objectives). Moreover, in addition, more emphasis is placed on avoiding limit reference points in decision-making rather than targeting optimal ones (Caddy and Mahon, 1995; Jennings, 2005; ICES, 2008b).

To define limit reference levels, we considered five axes in the definition of ecosystem overexploitation (Table 2): significant decline in biological diversity, significant increase in interannual variation, significant decrease in resistance of the ecosystem to perturbations, significant modification of the trophic structure of the ecosystem, and significant reduction in fish size. Although different types of ecosystem were scrutinized, interesting patterns emerged from the expert survey. First, there was no significant difference in reference levels provided for upwelling and temperate ecosystems (Table 3, Figures 5 and 7). This result has a direct consequence on the use of pie diagrams, which can then be considered a sensible way of representing and comparing ecosystem states by adopting the same scales and indicators across ecosystem types. A difference was noticed for high-latitude ecosystems (Figure 7), but the size of the sample for that type of ecosystem was not large enough to be conclusive. Second, the distributions of reference levels emerging from independent questionnaires were rather unimodal (Figure 4), suggesting some expert scientific consensus at least in defining a range of values for limit reference levels. This was not the case, however, for the proportion of under- to moderately exploited species for which the distribution of limit reference levels provided is rather flat. The diversity of the responses may be because it is difficult to define a limit reference level for this indicator on an ecological basis only, because it is also a management decision and a conservation issue whether or not to accept that some species are overexploited. The reason why the exercise is more ambiguous for this indicator than for others in the suite may be linked to the fact that all species are considered equally and are not associated with any specific functional role. For example, predatory fish or long-lived species are assumed to play a dampening role in ecosystem dynamics (Sala, 2006), and if overfished, can lead to trophic cascades (Frank *et al.*, 2005, 2006; Daskalov *et al.*, 2007).

On the other hand, from the comments given by the experts for justifying their choice, it has to be reported that for some indicators (and not only for the proportion of under- to moderately exploited species), it was difficult to associate clear ecosystem processes with the limit reference levels provided. Although it is theoretically straightforward to associate the significance of a limit threshold for TL of the landings, mean length, or mean lifespan with fishing effects, e.g. the ecosystem is considered overexploited if the fish community becomes dominated by small and short-lived prey fish, it is more difficult to associate a threshold of $1/CV$ of biomass to a particular level of fishing considered harmful to the ecosystem. Many limit reference levels were provided by experts in the form of relative values (as opposed to absolute values), based on time-series of indicators observed in the ecosystem considered or in similar types of ecosystem. Experts referred to known situations (past or present) where fishing was recognized to have had deleterious effects on ecosystems. As stated earlier, this way of defining limit reference levels was proposed as an alternative to an expert survey and was adopted in 53.3% of the answers for TL of the landings, 50% for mean length of the community, 44.1% for mean lifespan, 37.1% for proportion of predators, 36.7% for $1/CV$ of total biomass, and 22.2% for proportion of under- to moderately exploited species. This alternative was inspired by the work of Link (2005), who defined limit reference points for a set of ecosystem indicators based on experience in the Georges Bank and Gulf of Maine ecosystem, where the limit reference points corresponded to empirical observations after periods of intense fishing pressure. Therefore, the survey undertaken here was informative because it confirmed the utility of such a Delphi-empirical approach. The emergence of modal reference levels for most indicators encourages further refinement, and at least provides food for thought to stimulate the development of alternative approaches. By showing no significant difference between the reference levels provided for different types of ecosystem, the results of this study confirm that visual comparison of ecosystem state using the same pie diagram representations is justified. However, the results should be used cautiously because there is great dispersion around the modes of reference levels (Figure 4) and, compared with single-species reference levels, they are not drawn from such a well-established theoretical and practical background. It needs to be mentioned here too that only half the questionnaires sent were returned; many of the non-respondents (who were preselected for their supposed expertise in ecosystem indicators) felt that they did not have the necessary, complete knowledge on ecosystem indicators and related conceptual models of ecosystem functioning to complete the questionnaire fairly.

Comparing the ecological states of exploited marine ecosystems

Representation of the six ecological indicators by pie diagrams allowed us to identify ecosystems that appear to be (currently) more heavily affected by fishing (i.e. at least half the segments at minimum values and the others at low-to-medium values): northern Humboldt, north-central Adriatic Sea, southern Catalan Sea, Bay of Biscay, and Baltic Sea (Figure 2). The same ecosystems plus Sahara coastal (Morocco) and the Portuguese EEZ have at least half their indicator values below the modal limit reference levels provided by the expert survey. This simple evaluation and comparative approach is in line with local knowledge of fishing pressure and impacts on ecosystems. Both the southern Catalan Sea and the north-central Adriatic Sea have undergone a long history of exploitation, and current levels of fishing are high (Sardà, 1998; Jukic-Peladic *et al.*, 2001; Coll *et al.*, 2008b). The Baltic Sea faced an ecosystem regime shift at the turn of the 1980s/1990s when the cod-dominated system was replaced by clupeids. This significantly influenced both the structure and the functioning of the ecosystem, including the status of fish resources (Möllmann *et al.*, 2009). In the years 2003–2005, the period reported in the present study, the Baltic ecosystem was still very impacted by fishing, all indicators having relatively low values. In contrast, as discussed by Shannon *et al.* (2010), the situation with the northern Humboldt ecosystem was not well encapsulated by the comparative approach because it is an ecosystem that in recent years was dominated by anchoveta (*Engraulis ringens*; Bertrand *et al.*, 2004). It appears to be in a poor state relative to several other of the ecosystems presented, but this is not completely true. Although there are indications that Peruvian hake (*Merluccius gayi*) are overexploited (Ballón *et al.*, 2008; Guevara-Carrasco and Lleonart, 2008; Marzloff *et al.*, 2009), the Peruvian anchovy fishery is still flourishing (Chavez *et al.*, 2008). The dominance of anchovy in the biomass and catches of the Peruvian ecosystem tends to lower all indicators, except the proportion of under- to moderately exploited species. Therefore, comparison of this ecosystem with others should be undertaken with caution because the signals potentially captured by the ecosystem indicators are largely influenced by the state and dynamics of the anchovy population.

This situation parallels the potential problems encountered when looking at population indicators: some populations have great variability in recruitment, tending to counter expected trends in indicators of fishing effects. A decrease in mean length of a population does not necessarily reflect an increase in fishing pressure, but can be due to an increase in recruitment (Shin *et al.*, 2005; Bundy *et al.*, 2010). Likewise, a decrease in mean length of a community can be due to a high biomass of a small pelagic fish such as anchovy. Therefore, the set of ecosystem indicators proposed here needs to be interpreted carefully when dealing with systems that are clearly dominated by small pelagic species. It should also be borne in mind that the pie diagrams represent ecosystem states in a comparative framework. Hence, even those ecosystems which appear to be less affected by fishing are subject to intense exploitation. For example, on the eastern Scotian Shelf, there was a basic shift in the structure, functioning, and species composition in the early 1990s (Bundy, 2005), from a large fish, demersally dominated system to one dominated by small pelagic fish species and invertebrates.

Results from the present study are consistent with more-complex comparative studies. Using a decision-tree approach with associated decision rules to classify the 19 ecosystems into "good(ish), bad and ugly", based on significant trends in the six trend indicators, Bundy *et al.* (2010) reached similar conclusions for the period 1996–2005. However, they classified the Irish Sea as "ugly", or deteriorating, based on a significant decline in TL of landings from 1996 to 2005, whereas the results presented here suggest that the Irish Sea is one of the least affected of the ecosystems investigated. Similarly, the southern Catalan Sea was classified as "good(ish)" or improving by Bundy *et al.* (2010), whereas it was considered one of the ecosystems most affected by fishing in this work. These results are not inconsistent though, because the southern Catalan Sea was classified by Bundy *et al.* (2010) as "ugly" in the preceding period (1980–2005); it is now improving after a long history of exploitation. In the examination of the world's fisheries and fished systems by Worm *et al.* (2009), there was significant overlap with the list of ecosystems and species considered in this paper (Shin *et al.*, 2010). Comparing the rating of the systems between the two studies, there is a high level of concurrence (around 70%), especially when considering multispecies indices. Indeed, there is only disagreement for South American and Mediterranean ecosystems, for which Worm *et al.* (2009) relied solely on catch data. Agreement between the studies is much lower (at roughly 40%) when based on single-species information from stock assessments, which fail to capture the broader range of information in well-studied places like the North Sea or the Northeast United States. Interestingly, assessing systems only from single-species assessments as done by Worm *et al.* (2009) seems generally to produce a more pessimistic estimate of the situation. This is probably a reflection of the fact that those species that undergo intensive quantitative assessments are either vulnerable or primary target species (and therefore under potentially significant pressure), and neglects functional redundancies of the system (Auster and Link, 2009). Consequently, a potentially biased view of the overall system may be produced by a stock focus alone, lending support for the use of broader suites of indicators and integrative multispecies indicators for assessing fishing effects in ecosystems.

Conclusions

The approach presented here is a simple but informative one that ultimately aims to provide a way to communicate easily the states of exploited marine ecosystems to the general public (www.indiseas.org). Some care has been taken to reduce potential biases that may arise when comparing the states of ecosystems, but clearly this approach is a first step that needs to be complemented by more complex and integrated analyses (e.g. Bundy *et al.*, 2010; Coll *et al.*, 2010), by analyses of recent trends (e.g. Blanchard *et al.*, 2010), by consideration of local knowledge on ecosystem functioning (e.g. Shannon *et al.*, 2010), and by other potential drivers (e.g. Coll *et al.*, 2010; Link *et al.*, 2010). The scientific expert survey dedicated to the definition of limit reference levels and the meta-analysis of the information from that survey show potential for further refinement in the quest for classification and quantification of the impacts of fishing in marine ecosystems worldwide. Finally, the study shows that, even when based on simple indicators, a comparative approach has heavy requirements in terms of data collection, analyses, and standardizations (Jouffre *et al.*, 2010), and these cannot be met without the participation of local experts in the study.

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