The good(ish), the bad, and the ugly: a tripartite classification of ecosystem trends

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Bundy, A., Shannon, L. J., Rochet, M-J., Neira, S., Shin, Y-J., Hill, L., and Aydin, K. 2010. The good(ish), the bad, and the ugly: a tripartite classification of ecosystem trends. – ICES Journal of Marine Science, 67: 745–768.

Marine ecosystems have been exploited for a long time, growing increasingly vulnerable to collapse and irreversible change. How do we know when an ecosystem may be in danger? A measure of the status of individual stocks is only a partial gauge of its status, and does not include changes at the broader ecosystem level, to non-commercial species or to its structure or functioning. Six ecosystem indicators measuring trends over time were collated for 19 ecosystems, corresponding to four ecological attributes: resource potential, ecosystem structure and functioning, conservation of functional biodiversity, and ecosystem stability and resistance to perturbations. We explored the use of a decision-tree approach, a definition of initial ecosystem state (impacted or non-impacted), and the trends in the ecosystem indicators to classify the ecosystems into improving, stationary, and deteriorating. Ecosystem experts classified all ecosystems as impacted at the time of their initial state. Of these, 15 were diagnosed as "ugly", because they had deteriorated from an already impacted state. Several also exhibited specific combinations of trends indicating "fishing down the foodweb", reduction in size structure, reduction in diversity and stability, and changed productivity. The classification provides an initial evaluation for scientists, resource managers, stakeholders, and the general public of the concerning status of ecosystems globally.

Keywords: comparative approach, decision tree, ecosystem classification, ecosystem indicator, exploited marine ecosystems.

Received 11 June 2009; accepted 18 November 2009; advance access publication 6 January 2010.

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Introduction

Marine ecosystems have been subjected to anthropogenic forcing since humans first learned how to fish many thousands of years ago (Jackson *et al.*, 2001; Lotze and Milewski, 2004; Lotze *et al.*, 2006). That pressure has grown to the extent where serious concern is being expressed about the health of the world's ecosystems (Hollingworth, 2000; Jackson *et al.*, 2001; Pauly *et al.*, 2005; Coll *et al.*, 2008a). Indeed, a recent study has shown that there is now barely any part of the world's oceans that has not been affected at some level by anthropogenic activity, be it fishing, pollution, shipping, or eutrophication (Halpern *et al.*, 2008). In addition, we are living through a period of significant environmental change, the effects of which we are only beginning to explore (Hays *et al.*, 2005; Bender, 2007; ICES, 2008a; Cheung *et al.*, 2009), and which are difficult to predict.

From a fisheries perspective, the traditional approach of single-species stock assessment and management is being replaced by a more holistic ecosystem approach to fisheries, EAF (or variations on that theme; FAO, 2003; Garcia *et al.*, 2003; Daan *et al.*, 2005; Pitcher *et al.*, 2009). EAF still includes single-species stock assessment, but is expanded to include the wider impacts of fishing on the ecosystem, the role of the environment on species and ecosystem dynamics, the impacts of other activities, and the engagement of stakeholders in the processes leading to decision-making (Rice, 2008). The response of the fisheries scientific community has been to develop tools to facilitate an EAF, a basic component of which is the development of ecosystem indicators (Daan *et al.*, 2005), to evaluate the status and dynamics of ecosystems, or components thereof.

One way to use ecosystem indicators in management is to link them to (i) clear objectives, i.e. what is to be achieved, (ii) reference points or reference trends, i.e. measures of management performance, and (iii) control rules, i.e. actions required when management does not meet objectives (FAO, 2003; Cury *et al.*, 2005a). Ecosystem-based objectives, reference points, and control rules are difficult to set because of a lack of theory or because of limitations in the understanding of ecological complexity, uncertainties in data quality and model behaviour, and difficulties in balancing multiple and conflicting stakeholder interests (Cury *et al.*, 2005b). However, methods are being developed to overcome these difficulties, such as using historical or theoretical patterns to define reference points for indicators (e.g. Jennings and Blanchard, 2004). Jennings and Dulvy (2005) and Trenkel *et al.* (2007) argue that knowledge of the direction of trends in ecological indicators (specifically in size-based indicators) can be sufficient to support management decision-making. This means that there may be no need to identify absolute reference points for such indicators (Jennings and Dulvy, 2005); the more essential action is to stop the trend and to reverse it.

It is generally agreed that there is no single indicator that can provide management with all the information required for an EAF (FAO, 2003; Rice, 2003; Jennings, 2005). Rather, a suite of indicators that captures a range of impacts on the various attributes of the ecosystem and their response is required, with the results synthesized or integrated through means such as traffic-light analysis (Koeller *et al.*, 2000; Halliday *et al.*, 2001; Caddy, 2002), multivariate methods (Link *et al.*, 2002, 2010; DFO, 2003; Coll *et al.*, 2010), or a decision-tree approach (Rochet *et al.*, 2005; Jarre *et al.*, 2006; Trenkel *et al.*, 2007).

Multicriteria decision analysis (MCDA; Keeney and Raiffa, 1976; Belton and Stewart, 2002) has a wide and varied application in fisheries and resource management (McDaniels, 1995; Mardle and Pascoe, 1999; Paterson *et al.*, 2007; Jarre *et al.*, 2008). It is particularly useful for integrating different types of information and reconciling different objectives among stakeholders with diverse interests (Keeney and Raiffa, 1976). Generally, it is used as an aid to decision-makers, to analyse the components of a problem, and to compare the choices that can be made to discuss trade-offs. MCDA methods include decision trees, essentially models that portray the problem as a tree, with the overall problem or objective at the top, and its various subcomponents on the branches of the tree (FAO, 2003). In a classic application of the decision tree, each branch would terminate in a specific objective and related indicator (Keeney and Raiffa, 1976; FAO, 2003). Different methods can be applied to move back from the leaves (indicators) to the stem (overall objective), e.g. rule-based or fuzzy logic (see, e.g., Jarre *et al.*, 2008, for a comparison of these two popular methods). Rochet *et al.* (2005) developed a rule-based decision-tree approach to classifying marine ecosystems into improving, stationary, or deteriorating, based on the trends of key population and community indicators. That approach is adapted here to explore the use of a decision-tree approach for the diagnosis of a broad range of ecosystems with respect to their initial condition (unimpacted or impacted), using a suite of ecosystem indicators derived from commercial catch data and fisheries-independent survey data.

The suite of ecosystem indicators estimated for the 19 ecosystems included in this analysis was selected and calculated by the IndiSeas WG, which was established under the auspices of the EurOceans European Network of Excellence, to look at "EAF indicators: a comparative approach across ecosystems". This paper is one of a suite of papers published together that uses a comparative approach to evaluate the effects of fishing on marine ecosystems (Shin and Shannon, 2010; Shin *et al.*, 2010; and other sister papers in this suite). A comparative approach is particularly useful in an ecosystem context because the multiple ecosystems can be treated as pseudo-replicates. It therefore provides more confidence in the observed patterns of response to fishing seen across multiple ecosystems, rather than as observations in one ecosystem alone.

Methods

The ingredients for the decision tree in this analysis are a definition of the initial state of the ecosystem(s), the selection of a suite of ecosystem indicators for one or more ecosystems, an analysis of the trends of the ecosystem indicators, and the development of a decision tree with decision rules.

The 19 ecosystems examined cover a broad geographic range, including the eastern Pacific (north, central, and south); the Northwest Atlantic; the eastern Atlantic (north, central, and south), and the Mediterranean Sea (Table 1). They include high latitude, temperate, tropical, and upwelling systems, are associated with developing and developed nations, and have varied fishing histories. Moreover, they include ecosystems that are not normally included in such comparisons owing to a lack of data. The latter constrains the choice of indicator (see below), but it expands the scope of the comparison and the value of this study. Therefore, they offer a varied group of ecosystems and span a range of exploitation levels for classification purposes. A description of each ecosystem is provided in Shin *et al.* (2010).

Definition of initial state

To interpret the results of the decision tree, it was necessary first to describe the state of the ecosystem at the beginning of the period under consideration. Initial ecosystem state was simplified to a binary state: unimpacted or impacted. This was determined by asking the experts representing each of the 19 ecosystems to complete a survey in which three criteria were used to assess its state. An ecosystem was defined as impacted if two or more of the following criteria applied at the beginning of the time-series: Criterion (1) the proportion of fully/overexploited stocks >0 (i.e. stocks were already fully or overexploited at the beginning of the time-series), (2) industrialized or destructive fishing practices were already prevalent in the area (e.g. trawling over hard substrata, dredging, dynamite, discarding, cyanide fishing, or blast fishing), and (3) there were documented community or ecosystem impacts caused by fishing, such as habitat loss, impact on bycatch species, disruption of the foodweb, or loss of top predators. Respondents were asked to provide references to support their assessment (see Appendix A for more detail).

The suite of ecosystem indicators

The suite of ecosystem indicators developed by the IndiSeas WG included the following: mean length of fish in the ecosystem, mean lifespan of fish in the ecosystem, total biomass of surveyed species in the ecosystem, proportion of predatory fish in the ecosystem, trophic level of landings, and inverse fishing pressure (Table 2). These six trend indicators and the rationale associated with their selection are fully described in Shin *et al.* (2010). Data for the indicators were derived primarily from fisheries-independent surveys and commercial catch data, with auxiliary information used as indicated in Table A1 of Shin *et al.* (2010). Each indicator is specifically associated with one of the four ecosystem attributes; resource potential, ecosystem structure and functioning, conservation of functional biodiversity, and ecosystem stability and resistance to perturbations (Table 2; for further detail, see Shin *et al.*, 2010).

Rochet *et al.* (2005) used a combination of two population and six community indicators for their diagnostic analysis, although they explicitly used just community indicators for their decision tree. In contrast, ecosystem indicators, representing invertebrates,

Table 1. A list of the 19 ecosystems considered in the decision-tree classification of ecosystems.

Coastal ecosystem	Geographical area	Adjacent countries	FAO fishing zones	First year of time-series	Last year of time-series
Bering Sea, Aleutian Islands	Northeast Pacific	Alaska, USA	MFA: 67	1978	2005
West coast Canada	East Central Pacific	Canada		1980	2005
Southern Humboldt	Southeast Pacific	Chile	MFA: 87, Div: 2.2.	1993	2004
Northern Humboldt	Southeast Pacific	Peru	MFA: 87, Div: 1.1, 1.2	1983	2005
Eastern Scotian Shelf	Northwest Atlantic	Canada	MFA: 21, Div: 4V, 4W	1970	2005
Northeast United States	Northwest Atlantic	United States		1963	2005
Bay of Biscay	Northeast Atlantic	France	MFA: 27, Div: VIIIa, b	1994	2005
North Sea	Northeast Atlantic	UK, Norway, Denmark, Germany, Netherlands, Belgium	MFA: 27, Div: IVa,b,c, IIIa	1983	2005
Barents Sea	Northeast Atlantic	Norway	MFA: 27, Div: I, IIb	1984	2005
Irish Sea	Northeast Atlantic	Ireland, UK	MFA: 27, Div: VIIa	1973	2005
Portuguese EEZ	Northeast Atlantic	Portugal	MFA: 27, Div: IXa	1981	2005
Baltic Sea	Northeast Atlantic	Germany, Estonia, Sweden, Poland, Russia, Lithuania, Latvia, Finland	MFA: 27, Div: IIId 25 to 29	1974	2005
Southern Catalan Sea	Northwest Mediterranean	Spain	MFA: 37, Div: 1.1	1978	2003
North-central Adriatic Sea	Central Mediterranean	Italy, Slovenia, Croatia, Bosnia–Herzegovina, Montenegro	MFA: 37, Div: 2.1	1976	2005
Sahara coastal–Morocco	East Central Atlantic	Morocco	MFA: 34, Div: 1.3	1998	2005
Senegalese EEZ	East Central Atlantic	Senegal	MFA: 34, Div: 3.12	1986	2005
Guinean EEZ	East Central Atlantic	Guinea	MFA: 34, Div: 3.13	1985	2005
Mauritanian EEZ	East Central Atlantic	Mauritania	–	1982	2005
Southern Benguela	Southeast Atlantic	South Africa	MFA: 47, Div: 1.6, 2.1	1986	2005

MFA, FAO Major Fishing Area; Div, FAO Division (adapted from Shin *et al.*, 2010), including the years for which data are available to estimate the ecosystem indicators.

demersal fish, and pelagic fish, are used here (Table 2). The IndiSeas WG uses the term ecosystem indicator, whereas others such as Rochet *et al.* (2005) used the term community indicator. We use the wider term for two reasons: (i) the indicators encompass more than one community (minimally demersal and pelagic fish), and (ii) each indicator is specifically associated with one of the four ecosystem attributes listed above. Blanchard *et al.* (2010) show that the six indicators provide complementary information on the status of marine ecosystems (see below).

A comparative approach across diverse ecosystems requires that data for the indicators are available and operationally equivalent for all systems compared. This constrains the choice of indicators (Shin *et al.*, 2010), and in some cases, the length of the time-series. The length of the time-series of indicators for each ecosystem varies from a low of 8 years (Sahara coastal–Morocco) to a high of 43 years (Northeast United States), with a mean of 24 years (Table 1). For four of the ecosystems studied, one indicator was missing from the suite of metrics: Sahara coastal–Morocco, West coast Canada, and northern Humboldt lacked fish size, and Bay of Biscay lacked mean lifespan.

Determination of indicator trends

The information used in the decision tree is the trend of the indicators over the period being considered. For each indicator, we assume that a negative trend indicates increasing impacts of fishing (Table 2; see Shin *et al.*, 2010), whereas a positive trend indicates an improving situation. No trend indicates that there are either stationary impacts, meaning that the system remained in an unchanged impacted state, so the indicator did not change, or there were no detectable impacts of fishing. The initial state of the ecosystem determines the interpretation of trends. An impacted state can improve or deteriorate further, whereas an unimpacted ecosystem cannot be classified as improving because, by definition, there is no impacted state to improve upon (in relation to fisheries impacts).

Trends in the indicators were examined following the method described in Coll *et al.* (2008a) and used in Blanchard *et al.* (2010), using a two-stage estimation procedure, correcting for autocorrelation if present. To do this, a linear model was fitted to each of the predicted time-series using a generalized least-squares regression framework that models the temporal correlations in the error. The significance of the trend was assessed by testing the null hypothesis that the slope of the fitted line equals zero, using a two-tailed test of significance. A two-tailed test was used because, although we are interested primarily in testing the effects of fishing, we were also testing for potential recovery from the effects of fishing (positive trends in the indicators).

The decision tree and indicator order

The indicators in the decision tree are assessed sequentially and decision rules developed to integrate the information to classify the ecosystem. For their community indices, Rochet *et al.* (2005) used a conservative, precautionary decision rule where, as soon as one indicator had a significant negative trend, the decision tree was stopped and the ecosystem classified as deteriorating. For an ecosystem to be classified as improving, two indicators had to show a significant increase. The first rule took precedence over the second. In the decision tree, therefore, the trend of each indicator is sequentially assessed until either a negative trend is encountered or the end of the decision tree is reached (see

Table 2. Suite of six ecosystem indicators for diagnosing ecosystem status with corresponding ecosystem attribute.

Indicator	Headline label	Source	Calculation, notations (units)	Expected trend under fishing pressure	Ecosystem attribute
Mean length of fish in the community	Fish size	Fisheries-independent surveys	$\bar{L} = \sum_i L_i / \sum_i N_i$ (cm)	Decline	Ecosystem structure and functioning
Total biomass of species in the community	Biomass	Fisheries-independent surveys	B_s (t)	Decline	Resource potential
Mean lifespan of fish in the community	Lifespan	Fisheries-independent surveys	$\bar{A}_{\max} = \sum_s (A_{s,\max} B_s) / \sum_s B_s$ (years)	Decline	Ecosystem stability and resistance to perturbations
Proportion of predatory[a] fish in the community	% Predators	Fisheries-independent surveys	Proportion of predatory fish = biomass of predatory fish/biomass surveyed	Decline	Conservation of functional biodiversity
Trophic level of landings	Trophic level	Commercial landings and estimates of trophic level (empirical and fishbase)	$\overline{\mathrm{TL}} = \sum_s \mathrm{TL}_s Y_s / Y_{\mathrm{Total}}$	Decline	Ecosystem structure and functioning
Survey biomass/landings	Inverse fishing pressure[b]	Commercial landings and fisheries-independent surveys	B_{rs}/Y retained species[c]	Decline	Resource potential

N_s, total abundance of sampled species estimated from the research survey (as opposed to species sampled in catches by fishing vessels), including species of demersal and pelagic fish (bony and cartilaginous, small and large), as well as commercially important invertebrates (squids, crabs, shrimps, etc.). Intertidal and subtidal crustaceans and molluscs, such as abalones and mussels, mammalian and avian top predators, and turtles, are excluded. Surveyed species are those that are considered by default in the calculation of all survey-based indicators. L, length (cm); i, individual; s, species; N, total abundance; B, total biomass; Y, landings (t); $A_{\max}$, the mean lifespan of the community; $A_{s,\max}$, species-specific maximum expected age; Y_s and Y_{total}, landings of each species and total landings, respectively; TL_s, trophic level of each species; TL, mean trophic level of landings (adapted from Shin *et al.*, 2010); B_s, biomass of species sampled by researchers during routine surveys; B_{rs}, biomass of surveyed species retained by the fishery.

[a]Predatory fish are considered to be all fish species surveyed that are not largely planktivorous (i.e. phytoplankton and zooplankton feeders excluded). A fish species is classified as predatory if it is piscivorous or if it feeds on invertebrates that are larger than the macrozooplankton category (>2 cm). Detritivores are not classified as predatory fish.

[b]Fishing pressure is inverted so that it will decrease under increasing fishing pressure, so theoretically varying in the same direction as the other indicators in the selected suite.

[c]Retained species are the species caught in fishing operations and landed. Retained species are those that are considered by default in the calculation of all catch-based indicators.

Figure 2 of Rochet *et al*., 2005). This is a precautionary approach which insists that one negative trend in an ecosystem indicator intimates that the ecosystem is deteriorating and that this should invoke some type of remedial management response. The requirement that two indicators must increase to be classified as improving is again conservative. Both are somewhat *ad hoc* and would require agreement between decision-makers and stakeholders.

This "one strike and you are out" rule was adopted here as Decision Rule 1, but other decision rules were also explored in the analysis. A second, less precautionary, but more conservative rule "two strikes and you are out" was developed where a deteriorating state was reached when two indicators had a negative trend and improving when there were no negative trends and three indicators had a positive trend. As in the previous analysis, this still meant that classification of an ecosystem as improving was conservatively undertaken.

The initial order of the indicators was determined following Rochet *et al*. (2005), who reasoned that the ecosystem indicators estimated with the best precision and which were most clearly associated with interpreting a trend entered the decision tree first. A third criterion was added to this logic: each of the ecosystem attributes outlined in Table 2 had to be represented in the first four indicators. To this end, the order depicted in Table 2 was used. However, owing to the precedence of negative trends in the decision rules used here, the order of the indicators will not influence the final classification.

Exploring changes in ecosystem status over time

The time window over which data are explored can affect interpretation of the results. To explore whether the results observed for these ecosystems are consistent over time, the data were further explored in three time-blocks: (i) 1960s/1970s to 2005 (the seven ecosystems with the necessary data; Table 1), (ii) 1980–2005 (16 ecosystems with the necessary data; Table 1; also see Coll *et al*., 2010), and (iii) 1996–2005 (all 19 ecosystems; Table 1; see Blanchard *et al*., 2010). Only the order given in Table 2 and Decision Rule 1 were used in the decision tree. Initial state in periods (ii) and (iii) was determined by the results of the decision tree for periods (i) and (ii), respectively, for ecosystems whose original initial state was defined in earlier years (Table 1).

Independence of indicators

One assumption of the decision-tree method is that the six indicators are independent. Blanchard *et al*. (2010) conducted two sets of tests to explore redundancy in the indicators using a pairwise correlation analysis and mutual information analysis, which compares the rhythms of two time-series to quantify the extent of their dynamic cohesion (Cazelles, 2004). Results indicated that there were no clear redundancies across all indicators and across all systems and that the classification of ecosystems into groups according to pairwise correlations of indicators resulted in fairly weak associations across the ecosystems. There is no universal pattern across ecosystems and indicators, so there is no basis to exclude any one indicator from the analyses. Hence, the assumption of independence between indicators is reasonable for the periods used here when all 19 ecosystems are considered. However, within-system indicator dependencies for individual systems may affect the interpretation of the decision trees and are discussed where relevant.

Results

The initial state of one ecosystem began in the 1960s, six in the 1970s, nine in the 1980s, and three in the 1990s. All ecosystems met Criterion 1, i.e. that the proportion of fully/overexploited stocks >0, and, for at least 50% of the ecosystems, the proportion was >0.5. All ecosystems had industrialized or destructive fishing practices, and all but four (Guinea, Mauritania, Portugal, and Senegal) had documented community or ecosystem impacts caused by fishing (Appendix A). Therefore, all 19 ecosystems were classified as impacted because they met two or more of the three criteria outlined above.

Determination of indicator trends

The analysis of trends in ecosystem indicators over the entire length of their time-series demonstrates that most of the systems were undergoing change (Table 3). Only the Bay of Biscay appeared to show no significant change in state; in that case, three positive and two negative indicator trends were observed, but none were statistically significant. West coast Canada had two positive trends (biomass and trophic level of the landings), and the Barents Sea and Bering Sea each had one positive trend (inverse fishing pressure, and fish size, respectively). For the Baltic Sea, Mauritania, Portugal, Senegalese EEZ, and the southern Humboldt, all significant trends were negative. A mixture of positive and negative significant trends in ecosystem indicators was found for the other ten ecosystems.

Decision-tree analysis

Using Decision Rule 1, one ecosystem was diagnosed as improving (West coast Canada; Figure 1a), three as not improving (Barents Sea, Bay of Biscay, and Bering Sea; Figure 1b), and the rest as deteriorating from their initial state (Figure 1c). West coast Canada improved from its initial state because two indicators, biomass and trophic level of landings, had positive trends, and no indicator had negative trends. Of the three non-improving ecosystems, one had no significant trends (Bay of Biscay), and the Barents Sea and the Bering Sea each had one positive trend (inverse fishing pressure and mean size, respectively). The ecosystems that were diagnosed as deteriorating from their initial state fell into four main groups: seven ecosystems were immediately classified as deteriorating because the first indicator at the top of the decision tree, mean length, decreased significantly (Figure 1c); three ecosystems (Portugal, Senegal, and the southern Humboldt) passed the first two levels of the decision tree, but showed significant decreases in their biomass (Figure 1c); Mauritania, the southern Benguela, and Sahara coastal–Morocco had significant decreases in the mean trophic level of landings (although biomass increased in the southern Benguela and Sahara coastal–Morocco; Figure 1c); and in the Guinean EEZ (Figure 1c), three indicators increased, but the last, inverse fishing pressure, decreased, eventually placing Guinea in the deteriorating class.

Decision Rule 2 yielded similar results, with the main difference that no ecosystem was classified as improving because none had three significant increases and no negative trends (Table 3), and four ecosystems were reclassified as not improving. Therefore, West coast Canada, which had been improving under Decision Rule 1, was classified as not improving because it only had two positive trends. In addition, Guinea EEZ, the southern Benguela, Portugal, and Sahara coastal–Morocco were reclassified as not improving because they had only one negative trend. However, 11 ecosystems remained classified as deteriorating.

Exploring changes in ecosystem status over time

Following Decision Rule 1, during the first period, six of the seven ecosystems were classified as deteriorating, and one as not improving. In the second period, 12 of the 16 ecosystems were diagnosed as deteriorating, 1 as not improving, and 3 as improving. In the last period (1996–2005), 8 of the 19 ecosystems were classified as deteriorating, 10 as not improving, and 1 as improving (Table 4).

For ecosystems spanning the three periods, diagnoses for three were consistent regardless of the period under consideration: the Irish Sea, the Baltic Sea, and the north-central Adriatic Sea were always diagnosed as deteriorating (Table 4, Appendix B). In other cases, the diagnosis changed. The eastern Scotian Shelf and the southern Catalan Sea were both defined as deteriorating until the mid-1990s, after which (1996–2005) the eastern Scotian Shelf reached some stability (no significant trends), and the southern Catalan Sea was classified as improving (increase in % predators and trophic level of landings; Appendix B). The Bering Sea changed from a stationary diagnosis to improving (positive trends in mean size and mean lifespan) to stationary over the three periods. Finally, the Northeast United States was diagnosed as deteriorating over the whole time-series, improving since the 1980s, and stationary since the mid-1990s.

The time-series of nine ecosystems began during the 1980s, and again, some diagnoses were consistent over time: Senegal, the southern Benguela, and the northern Humboldt were classified as deteriorating over both the 1980s to 2005, and 1996–2005 periods, whereas the Barents Sea was classified as not improving for both periods (Table 4). Four ecosystems, Guinea, Mauritania, North Sea, and Portugal, were diagnosed as deteriorating from the 1980s to 2005, but as not improving from 1996 to 2005, because there were no significant trends in any indicator, suggesting that, as with the eastern Scotian Shelf, they had reached stability. Lastly, west coast Canada was first diagnosed as improving (positive trends in biomass and trophic level of landings), but then as not improving for the most recent period.

The time-series of three ecosystems began in the early 1990s: two were diagnosed as deteriorating (Sahara coastal–Morocco, and the southern Humboldt), and the Bay of Biscay was classified as not improving (Table 4).

Discussion

One of the strengths of a comparative approach is that replication of observed patterns across ecosystems provides more confidence that such patterns are real than if they were observed in just one ecosystem. Two concerning patterns were replicated across the 19 ecosystems analysed here using a decision-tree approach: (i) all 19 ecosystems were considered impacted at the beginning of their time-series; (ii) since the initial state, 15 ecosystems

Table 3. Trends in ecosystem indicators (and in parenthesis their attributes) over the entire length of their time-series (to 2005) for the 19 ecosystems evaluated, using generalized least-squares and autoregressive error.

Coastal ecosystem	Fish size (EF)	Lifespan (SR)	Biomass (RP)	% Predators (CB)	Trophic level (EF)	Inverse fishing pressure (RP)	Length of time-series (years)
Baltic Sea	**−0.084**	−0.057	**−0.078**	−0.043	−0.047	−0.047	32
Barents Sea	0.072	0.053	0.087	−0.020	0.081	**0.070**	22
Bay of Biscay	0.063	–	−0.129	0.052	0.018	−0.027	12
Bering Sea, Aleutian Islands	**0.089**	0.001	0.022	0.023	0.038	−0.024	28[a]
Eastern Scotian Shelf	**−0.071**	**−0.050**	−0.041	−0.021	**−0.045**	**0.078**	36
Guinea EEZ	**0.106**	0.039	−0.010	**0.101**	**0.093**	**−0.124**	21
Irish Sea	**−0.118**	0.110	**0.043**	**0.046**	**−0.091**	**0.079**	33[b]
Mauritanian EEZ	0.075	0.044	0.045	0.010	**−0.144**	**−0.157**	24[c]
North-central Adriatic Sea	**−0.111**	**0.090**	**−0.069**	0.061	−0.033	**−0.070**	30[d]
Northeast United States	**−0.044**	**0.058**	**0.056**	**−0.050**	0.012	**−0.051**	43
North Sea	**−0.130**	0.010	**0.082**	0.038	**−0.065**	**0.145**	23
Northern Humboldt	–	**−0.119**	0.075	**0.085**	**0.133**	**−0.079**	23
Portuguese EEZ	0.000	0.033	**−0.068**	0.032	0.065	0.040	25
Sahara coastal–Morocco	–	0.003	**0.366**	−0.230	**−0.310**	**0.321**	8
Senegalese EEZ	−0.054	0.039	**−0.132**	0.041	−0.082	**−0.141**	20
Southern Benguela	0.020	−0.078	**0.167**	−0.043	**−0.080**	**0.158**	20
Southern Catalan Sea	**−0.122**	**−0.089**	**0.051**	−0.046	0.025	**0.059**	26
Southern Humboldt	−0.055	0.066	**−0.193**	0.058	**−0.196**	0.063	12
West coast Canada	–	0.024	**0.064**	0.038	**0.102**	0.020	26
Number of significant negative trends	7	3	5	1	7	6	
Number of significant positive trends	2	2	7	3	3	7	

Significance levels ($\alpha = 0.05$) are shown emboldened; –, no data.
EF, ecosystem structure and functioning; SR, ecosystem stability and resistance to perturbations; RP, resource potential; CB, conservation of functional biodiversity.
[a]Time-series for fish size was 24 years long.
[b]Time-series for fish size and lifespan was 16 years long.
[c]Time-series for fish size was 20 years long, and trophic level and inverse fishing pressure series were 16 years long.
[d]Time-series for fish size, lifespan, and % predators was 24 years long.

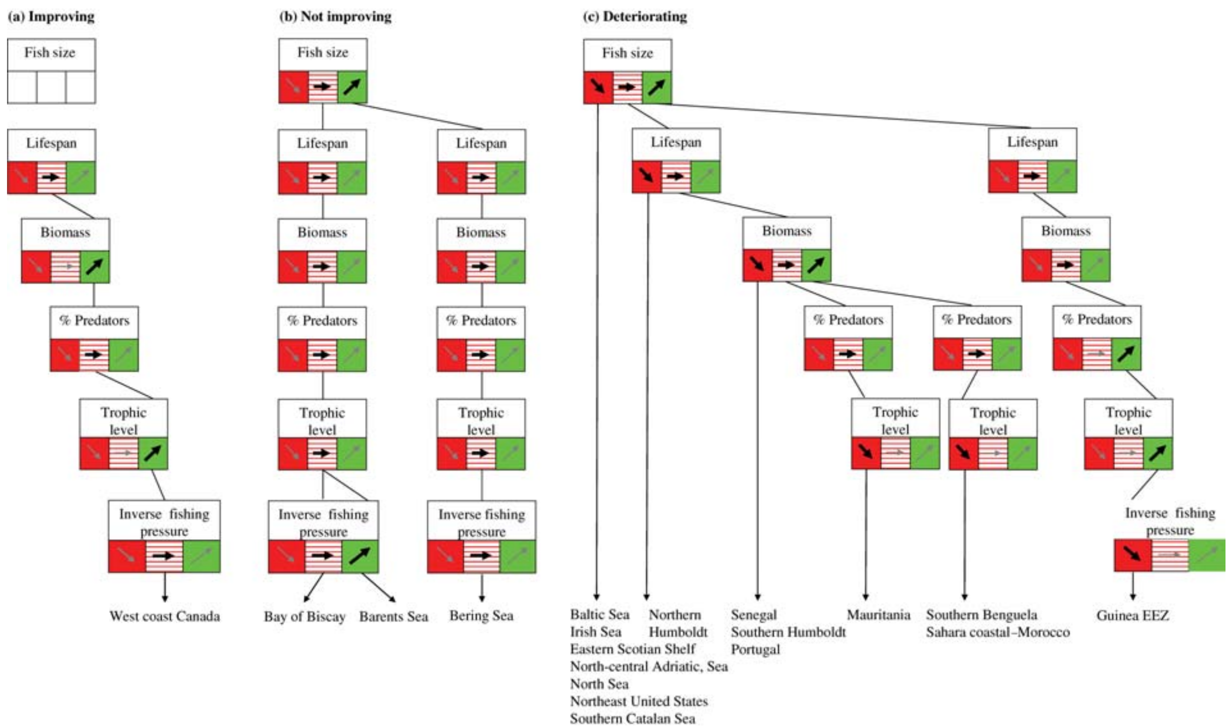

Figure 1. Decision-tree diagnoses for the 19 ecosystems evaluated: (a) improving, (b) not improving, and (c) deteriorating. The first node of the decision tree is fish size. If that is decreasing, Decision Rule 1 is invoked and the decision tree is terminated (c). If it is increasing or there is no trend, then the decision tree moves down to the next node (indicator), and the same process is followed until either Decision Rule 1 terminates the decision tree or the last node, inverse fishing pressure, is reached. The three arrows at each node indicate the three possible directions of the indicator trend, decreasing (red), increasing (green), or no trend (striped); the heavy black arrow indicates the observed trend. In (a), white nodes indicates missing data, where it is assumed that there is no change. See text for further detail.

deteriorated further under Decision Rule 1, and 11 under Decision Rule 2. This is not entirely unexpected because many of these ecosystems have exploitation histories extending over several hundreds of years, particularly in the North Atlantic and Mediterranean, and fishing does impact the ecosystem (see the contributions in Hollingworth, 2000). However, the consistency of the diagnosis provided here is worrisome. Given that all ecosystems departed from a state that was already impacted, we have relabelled our diagnoses for this study as ugly (deteriorating), bad (not improving, because the ecosystem is remaining in its impacted state), and good(ish) (improving, because the direction is good, but the ecosystems are still likely to be highly impacted).

Essentially, our results indicate that most of the ecosystems studied are in a more impacted state now than they were at the beginning of their time-series, i.e. that they are in an ugly state. Moreover, eight of the ecosystems were still deteriorating when examined over the past 10 years (1996–2005), and five were stationary, having deteriorated, and therefore still in a bad state. Few of the ecosystems studied, therefore, can be considered in a good state during any of the periods analysed, regardless of whether they are temperate, tropical, upwelling, or high latitude (Shin *et al.*, 2010; Table 1). Exceptions are the two systems from the North Pacific (Bering Sea and West coast Canada), which were diagnosed as improving from 1980 to 2005, and stationary since then. In comparison, ecosystems in the North Atlantic and Mediterranean have an exploitation history extending over several hundred years and were generally diagnosed as deteriorating or not improving from a previously deteriorated state, excluding the Bay of Biscay and the Barents Sea. By any measure, therefore, the results of this analysis are deeply disturbing.

Assessment of initial state

The method for assessing the initial state of the ecosystems was conservative and demanded that a system could only be described as unimpacted if it failed to meet two or more criteria at the start of the time-series. The criteria were selected to include the effects of fishing on species and the ecosystem, and were intended to maximize comparability. Assessing the initial state of ecosystems 20+ years ago can be challenging, because there was less interest in the ecosystem effects of fishing at that time, and less documentation. For some ecosystems, there was also simply less information available. Information for the first and second criteria was available for most ecosystems, but the third criterion, documented community or ecosystem impacts caused by fishing, leaves room for error. In several cases, there was no documentation for the earlier period, so rendering this criterion impossible to assess: absence of documentation does not equate to lack of impact. This did not affect the assessment of initial state of any single ecosystem, given the binary classification used here. However, if a less conservative method was used, such as degree of impact, Criterion 3 would be more important.

Time-frames

The results of the analysis were sensitive to the time frame used and, for some systems, revealed an evolutionary process. The

Table 4. Comparison of decision-tree diagnoses for three time-windows, pre-1980s to 2005, 1980–2005, and 1996–2005, under Decision Rule 1.

Start of time-series	Ecosystem diagnosis	Pre-1980s-2005 I	Pre-1980s-2005 Not I	Pre-1980s-2005 D	1980–2005 I	1980–2005 Not I	1980–2005 D	1996–2005 I	1996–2005 Not I	1996–2005 D
Pre-1980s	Baltic Sea			■			■			■
	Bering Sea, Aleutian Islands		■		■				■	
	Eastern Scotian shelf			■			■		■	
	Irish Sea			■			■			■
	North-central Adriatic Sea			■			■			■
	Northeast US			■	■				■	
	Southern Catalan Sea			■			■	■		
1980s	Barents Sea	░	░	░		■			■	
	Guinea ZEE	░	░	░			■		■	
	Mauritania	░	░	░			■		■	
	North Sea	░	░	░			■		■	
	Northern Humboldt	░	░	░			■			■
	Portuguese ZEE	░	░	░			■		■	
	Senegalese ZEE	░	░	░			■			■
	Southern Benguela	░	░	░			■			■
	West coast Canada	░	░	░	■				■	
1990s	Bay of Biscay	░	░	░	░	░	░		■	
	Sahara coastal Morocco	░	░	░	░	░	░			■
	Southern Humboldt	░	░	░	░	░	░			■

Dark grey cells indicated diagnosis. Light grey cells reflect no data for the periods indicated. I, improving; Not I, not improving; D, deteriorating.

Northeast United States, for example, was originally diagnosed as deteriorating (1963–2005), then improving (1980–2005), and finally (1996–2005) as not improving. Overall, the shortest time-frame of 10 years, 1996–2005, produced the least number of significant trends (20), whereas the longer periods produced more than twice this number (see also Blanchard *et al.*, 2010). This result was expected because it has been suggested that at least 10–15 years of data are required to detect trends (Nicholson and Jennings, 2004), and Trenkel *et al.* (2007) even suggested 20+ years.

However, we believe that there is a second, process-orientated explanation for the lack of trends in the recent 10-year time-frame for some ecosystems. Many of the ecosystems were diagnosed as deteriorating over the longer period and by the mid-1990s had reached a new more impacted state, after which they remained relatively stable. This was certainly the case for the eastern Scotian Shelf (Bundy, 2005), and likely the case for the Northeast United States (Link *et al.*, 2002) and the North Sea (Mackinson and Daskalov, 2007) too.

Of greater concern is the result that 6 of the 12 ecosystems diagnosed as deteriorating during the period 1980–2005 were still diagnosed as deteriorating over the period 1995–2005. The Baltic Sea, the Irish Sea, and the North-central Adriatic Sea have a long history of fishing, but notably have fewer negative trends in the recent periods than during the period 1980–2005 (Appendix B), suggesting a decline in the rate of deterioration. Senegal also had fewer trends in the recent period. The southern Benguela and the northern Humboldt are upwelling systems in which environmental drivers can influence ecosystem dynamics, productivity, and the distribution of marine organisms (Shannon *et al.*, 2008, 2010). The ecosystem indicators must therefore be interpreted in that context: for the southern Benguela and the northern Humboldt, the diagnosed deterioration for the period 1995–2005 is more apparent than real (see below).

Detecting ecosystem effects through a combination of trends

A key component of decision-tree methodology is the interpretation of the trends in ecosystem indicators. Always, the expected impact of fishing is a decrease in the indicator. However, fishing does not take place in isolation, and other factors such as changes in environmental conditions, regime shifts, increased productivity, or increased resource recruitment may also effect a change in indicator trend. A list of factors that may influence the direction of the six ecosystem indicators are suggested in Table 5. Moreover, certain combinations of indicators, as proposed below, may be indicative of the specific effects of fishing or environmental forcing: (i) fishing down the foodweb (FDFW); (ii) reduction in diversity and stability; (iii) reduction in size structure; (iv) changes in productivity or recruitment. We explored the combinations of trends in the 19 ecosystems to determine whether particular processes could be hypothesized using these trend combinations, and to determine whether other factors were known that could contribute to the results.

(i) FDFW takes place when the average trophic level of the catch declines over time with a concurrent decrease in the total catch (Pauly *et al.*, 1998a, 2001). Total catch is not included as an indicator here, but we suggest that a combination of a significant negative trend in the trophic level of the landings along with a significant negative trend in biomass would indicate FDFW (a decrease or no change in the other ecosystem indicators would also be consistent with FDFW). This combination of indicators was only observed for the eastern Scotian Shelf (Table 6), which has a history of FDFW (Bundy, 2005).

(ii) Reduction of diversity and stability could be indicated by a decrease in % predators, indicating a loss in functional diversity (Hooper *et al.*, 2005), and mean lifespan, indicating a loss in stability. Stability would be reduced because longer-lived fish are generally larger and hence more resistant to perturbation, because they are more fecund than smaller fish and have more-viable eggs (Longhurst, 2002). A decrease or no change in the trophic level of the landings, fish size, or inverse fishing pressure would be consistent with this effect; no specific trend is expected for biomass. This effect was only observed in two ecosystems (Table 6): the Baltic Sea (significant decrease in lifespan and % predators, accompanied by a significant decrease in mean length and trophic level of the landings) and the southern Benguela (significant decrease in lifespan and % predators, accompanied by a significant decrease in the trophic level of the landings).

(iii) Reduction of size structure is the first potential indicator of size-selective effects of fishing where large fish are targeted (Stokes *et al.*, 1993; Sinclair *et al.*, 2002), and it was the most common effect observed. Here, it is represented by a decrease in mean size coupled with a negative trend in mean lifespan and/or trophic level of the landings and/or % predators. Alternatively, if fishing selectively targeted small fish, the opposite trends would be expected. There is no expected trend in biomass or inverse fishing pressure. Reduction in size structure was evident in the Baltic Sea, the Bering Sea, the eastern Scotian Shelf, the North Sea, and Guinea. The combination of indicators on the eastern Scotian Shelf and southern Catalan Sea (significant negative trends in fish size and lifespan) and on the eastern Scotian Shelf and the North Sea (significant negative trends in fish size and trophic level) suggest that large fish have been targeted and selectively removed from those ecosystems.

Table 5. Summary of reasons why the six ecosystem indicators may increase or decrease.

Indicator and basis	Reasons for increase	Reasons for decrease	Source
Mean fish size			
This indicator is based on individual length data at the community level	• Increase in abundance of large fish in dominant populations • Decrease in recruitment in dominant populations • Increase in mean growth rates as a result of favourable environmental conditions • Decrease in abundance of stocks of small fish • Increase in abundance of stocks of large fish • Regime shift towards a condition favouring large fish species	• Loss of larger fish from dominant populations • Strong recruitment success in dominant populations • Decrease in mean growth rates attributable to unfavourable environmental conditions • Increase in abundance of stocks of small fish • Decrease in abundance of stocks of large fish • Regime shift towards a condition favouring small pelagic fish	Shin *et al.* (2005); Rochet and Trenkel (2005); Chavez *et al.* (2003, 2008)
Mean lifespan			
This indicator is not based on individual data, but on life-history traits weighted by species biomass: the variations are meant to capture changes in species composition	• Increase in the biomass of species with high longevity • Decrease in the biomass of species with short lifespan • Regime shift towards a condition favouring large fish species	• Decrease in the biomass of species with high longevity • Increase in the biomass of species with short lifespan • Demographic explosions of invertebrates (e.g. cephalopods, crustaceans) • Regime shift towards a condition favouring small pelagic fish	–
Biomass			
Biomass is the total amount of surveyed fish species in the ecosystem, expressed either as a unit per time or area or as an absolute value for the ecosystem	• Reduced mortality caused by a decrease in fishing pressure • Good reproductive success leading to good recruitment as a result of a large spawning-stock biomass, favourable climatic conditions, etc. • Reduced predation or increased food supply, i.e. favourable trophic interactions	• Increased mortality—both fishing and natural (through predation, pollution, unfavourable environmental conditions, etc.). Fishing mortality can be of two types, direct and indirect. The first is the catch of target species and includes the unintended mortality of non-target species, bycatch species, and discarding. The second includes increased predation or competition propagated through the foodweb as a result of species changes caused by fishing • Poor reproductive success attributable to unfavourable environmental conditions, leading to poor recruitment • Increased predation, increased competition, or depleted food supply, i.e. unfavourable trophic interactions	Duplisea *et al.* (1997); Rice (2005); Rochet and Trenkel (2005); Rogers and Greenaway (2005)

Continued

Table 5. *Continued*

Indicator and basis	Reasons for increase	Reasons for decrease	Source
% Predators Predators are defined as all surveyed species that are not planktivorous and feed on fish or invertebrates larger than 2 cm. Variations in this indicator capture changes in the trophic structure of the ecosystem. If predatory fish merely increase in proportion to overall fish biomass, then no proportional change is seen	• Reduced fishing on predatory fish • Distributional changes of predators or prey • Increased availability of prey for predators • Competitive release • Regime shift towards a condition favouring large fish species	• Increased fishing pressure on predatory fish or their prey • Distributional changes of predators and/or prey • Reduced availability of prey • Competitive exclusion • Regime shift towards a condition favouring small pelagic fish	Rochet and Trenkel (2005)

Trophic level of landings
This indicator is a measure of the average trophic level of the species exploited by the fishery
	• Early stage of exploitation of fishery
	• Shift in fishing effort from lower to higher trophic level species
	• Decrease in productivity at lower trophic levels
	• Targeting of larger individuals at higher trophic levels within a species.
	• Regime shift towards a condition favouring large fish species
		• Fishing effort shifted to lower trophic levels through:
		  • decrease in biomass of higher trophic levels (fishing down the foodweb)
		  • preferential shift to more productive lower trophic levels (fishing through the foodweb)
		  • decrease in size composition of commercial species and consequent decrease in trophic feeding level
		  • increase in availability or market value of commercially valuable invertebrates
		  • regime shift towards a condition favouring small pelagic fish
			Pauly *et al*. (1998a, b); Caddy *et al*. (1998); Essington *et al*. (2006)

Inverse fishing pressure This indicator is a proportion, and change will be a result of change in landed catch, biomass, or both. Moreover, if both catch and biomass change in the same direction, exploitation rate may not change	May be due to a decrease in catch, an increase in biomass, or both	May be due to an increase in catch, a decrease in biomass, or both	
	Decrease in landed catch through: • decrease in catch of one or more species • change in management regulations (more restrictive) • decreased productivity • one or more stocks fully or overexploited • loss of larger fish from the population(s) being exploited • species or stocks becoming less aggregated • socio-economic disincentives (e.g. loss of market, increased fuel costs)	Increase in landed catch through: • increase in catch of one or more species • change in management regulations (less restrictive) • increased productivity, few or no restrictions on catch • increase in fishing efficiency or effort • expansion of fishing into new areas • newly exploited stock(s) • species/stocks becoming more aggregated • improved fish-finding equipment • socio-economic incentives (e.g. new markets, lower fuel costs, subsidies)	
	Increase in biomass (see biomass row above)	Decrease in biomass (see biomass row above)	

Table 6. Comparison of potential effects on the ecosystem during three time-windows, pre-1980s–2005, 1980–2005, and 1996–2005, under Decision Rule 1.

Start of time-series	Ecosystem	Pre-1980s-2005				1980–2005				1996–2005			
		FDFW	DS	SS	PR	FDFW	DS	SS	PR	FDFW	DS	SS	PR
Pre-1980s	Baltic Sea						■	■					
	Bering Sea, Aleutian Islands							■					
	Eastern Scotian shelf			■		■		■					
	Irish Sea				■				■				
	Northeast US				■								
	Southern Catalan Sea												
1980s	Southern Benguela	–	–	–	–						■		
	North Sea	–	–	–	–			■	■				
	Guinean EEZ	–	–	–	–			■					
	Portuguese EEZ	–	–	–	–								
1990s	Southern Humboldt	–	–	–	–	–	–	–	–				

FDFW, fishing down the foodweb; DS, reduction in diversity and stability; SS, reduction in size structure; PR, increased productivity or recruitment. Only ecosystems with the effects observed are shown. Dark grey cells indicate an effect, and light grey cells no data for the periods indicated.

Guinea and the Bering Sea exhibited the opposite effect, with an increase in fish size, coupled with an increase in the trophic level of the landings and % predators (Guinea) and mean lifespan (Bering Sea). This indicates that either small fish were being targeted or are less abundant for reasons such as a regime shift to conditions favouring large fish or a decrease in recruitment (Table 5). In the Bering Sea, for example, significant positive trends in fish size and lifespan were attributable in part to longer-lived flatfish, which experienced strong recruitment in the 1980s possibly as a consequence of beneficial climate conditions (NRC, 1996).

(iv) Increased productivity and increased recruitment are difficult to separate with this set of indicators: a combination of a decrease in fish size (increased productivity at lower trophic levels, increased recruitment, or both) with an increase in biomass could be symptomatic of increased productivity, recruitment, or both. As a working model, it is suggested that an increase in productivity or recruitment would be propagated up the foodweb through bottom-up processes, leading to increased productivity at all trophic levels or size groups. It would likely have transient impacts, with potential time-lags, on mean lifespan, % predators, trophic level of the landings, and inverse fishing pressure. The Northeast United States, the Irish Sea, and the North Sea all showed evidence of increased productivity or recruitment, which may be due to removal of larger predators (Table 5, and see below). However, environmentally induced ecosystem changes or regime shifts may drive changes in the productivity of ecosystems, reflected in non-fishery-related changes in some of our indicators. For example, for the southern Benguela, significant declines in the trophic level of the landed catch, % predatory fish, and in mean lifespan, and biomass increase reflected the environmentally induced, temporary upsurge of small pelagic fish in the early 2000s rather than fishing effects (Shannon *et al.*, 2010).

These four proposed ecosystem effects were observed in 11 of the 19 ecosystems, but not consistently over time: most striking is the lack of observed effects in the latest period, 1996–2005, consistent with fewer significant ecosystem trends (see above). These observed ecosystem effects support the diagnosis of the decision tree: of the 15 ecosystems diagnosed as deteriorating during the longer periods considered (pre-1980s to 2005 and 1980–2005), 6 exhibited size-selective effects of fishing, 1 experienced FDFW, 2 experienced reduction in diversity and stability, and 4 showed evidence of increased productivity. Some of these ecosystem trends were consistent over time: the Irish Sea exhibited increased productivity over the whole time-series and since 1980, whereas the eastern Scotian Shelf exhibited signs of size-selective fishing of large fish over these two periods and FDFW over the period 1980–2005. During the latter period, the Baltic Sea exhibited reduction in diversity and stability and size structure, and the North Sea exhibited a decrease in size structure and increased productivity or recruitment. Increased productivity or recruitment was observed in the Irish Sea, the Northeast United States, and the North Sea. These ecosystems have been heavily exploited for more than a century and are not showing any signs of improvement, other than reduced exploitation and an increase in biomass, because there seem to be more small fish (decrease in fish size). For the Irish Sea, the diagnosis indicates that it has continued to deteriorate.

Evidence of other trends and non-fisheries effects

In addition to the proposed trends associated with specific ecosystem effects, several other trends were apparent in the data. Some ecosystems exhibited contradictory trends. For example, four ecosystems (Irish Sea, North Sea, Sahara coastal Morocco, and southern Benguela) exhibited a significant negative trend in the trophic level coupled with an increase in biomass (Appendix B). Biomass may have increased (rather than decreased) as a consequence of, for example, release from predation pressure attributable to fishery removals of large fish or by environmentally driven good recruitment (Table 5, and see below). In the Northeast United States and North-central Adriatic Sea, mean lifespan increased, whereas fish size decreased. In the former, this may be because fisheries have, for some time, been catching larger organisms and many of the remaining organisms, not primarily targeted by fisheries, such as skates and some scorpionids, have longer lifespans (Link, 2007). In the north-central Adriatic Sea, fishing mainly targeted and depleted small pelagic fish, leaving longer-lived fish in the population. Recruitment of long-lived fish will result in a higher mean lifespan, but because their recruits are small, average size in the community will decline. Using data for 1995–2005, Blanchard *et al.* (2010, Table 5) found that there was redundancy between lifespan and fish size in only four ecosystems that did not include either Northeast United States or north-central Adriatic Sea. In the southern Humboldt, mean fish size increased, but the trophic level of landings decreased over the past decade. Arancibia and Neira (2005) found a significant decline in the mean trophic level of landings from the southern Humboldt for the period 1978–2002, but Neira (2008) subsequently determined that this decline was first caused by fishing through the foodweb, *sensu* Essington *et al*. (2006), and that real FDFW had been taking place in that system only since the mid-1990s.

Environmental drivers can have a strong influence on indicator trends and need to be taken into consideration when interpreting indicator trends and decision trees. Several of the ecosystems analysed here are upwelling ecosystems, subject to strong environmental drivers (Ryther, 1969; Chavez *et al*., 2003; Shannon *et al*., 2008), and they are discussed in detail by Shannon *et al*. (2010). In the northern Humboldt, for example, strong climatic, bottom-up forcing affects fish productivity on a large range of scales (Pauly and Tsukayama, 1987; Pauly *et al*., 1989; Chavez *et al*., 2008; Gutiérrez *et al*., 2008), and the environment rather than fishing pressure appears to be the main driver of ecosystem change (Bertrand *et al*., 2004, 2008; Shannon *et al*., 2008; Taylor *et al*., 2008). Of the indicators explored here, mean lifespan and inverse fishing pressure significantly decreased, whereas % predators and trophic level of the catch increased over the period 1980–2005, and over the period 1996–2005, mean lifespan decreased but biomass increased (there was no mean length indicator for the northern Humboldt). Mean lifespan likely decreased, and biomass increased as a result of factors associated with a change from a warmer period in the 1980s (*El Viejo*, *sensu* Chavez *et al*., 2003) to cooler, more-productive conditions in the 1990s (*La Vieja*, *sensu* Chavez *et al*., 2003). The short-lived anchoveta (*Engraulis ringens*) increased (full anchovy era, *sensu* Gutiérrez *et al*., 2008), larger pelagic predators such as hake (*Merluccius gayi*) declined through overfishing and adverse climate conditions (Ballón *et al*., 2008; Guevara-Carrasco and Lleonart, 2008), and the biomass of the jumbo squid (*Dosidicus gigas*), a short-lived predator, increased dramatically (Argüelles *et al*., 2008), so increasing % predators. For further detail, see Shannon *et al*. (2010). In this case, the classification of the northern Humboldt as deteriorating for the recent period, 1996–2005, should be questioned recognizing the influence of environmental change on the indicators as a consequence of a regime shift (Table 5). This emphasizes the need for the incorporation of environmental indicators into the decision-tree model.

The selection of ecosystem indicators

One objective of IndiSeas was to apply a suite of indicators to assess the status of ecosystems. The suite of ecosystem indicators used here measures mostly the species sampled by demersal trawl surveys (Jouffre *et al*., 2010) and found in commercial catch data, but the longer-term objective of IndiSeas is to expand this to encompass the whole ecosystem, within the constraints noted above.

Link *et al*. (2002), Caddy (2004), and others have recommended using a broad range of indicators to assess the status of ecosystems. Several groups working towards an implementation of an EAF have also noted the need to select indicators using a set of criteria that are germane for the management issues at hand in a given ecosystem (Degnbol and Jarre, 2004; Degnbol, 2005; Link, 2005; Rochet and Rice, 2005; Methratta and Link, 2006). However, when making comparative evaluations of marine ecosystems globally, a more limited set of indicators has to be accepted because the data need to be available for all ecosystems. The implication of this is a compromise between an ideal and a realistic set of indicators. Although it is not suggested that the set of indicators selected here is complete, it is proposed as a minimum set which we do consider meaningful. The rationale for the choice of indicators is discussed further in Shin *et al*. (2010).

A second class of indicators that monitor the state of the environment, especially for ecosystems with large environmental fluctuations such as upwelling systems, is required to incorporate an important additional ecosystem driver. This exploratory analysis clearly showed that for some upwelling systems, the effect of the environment, with its associated regime shifts, had strong impacts on the ecosystem indicators, which could be mistaken for effects of fishing. Minimally, therefore, environmental indicators with associated decision rules should be incorporated into the decision tree to indicate when the environment may be impacting other indicators.

Of the six indicators used in this exploratory study, fewer significant trends were found for mean lifespan and % predators, both of which were derived from fisheries survey information, suggesting that they are less sensitive to change than the other indicators which were catch-derived. Although we assumed that biomass would decrease in response to fishing, it increased in three systems for other reasons, demonstrating that it is a complex indicator and may not be ideal for the purposes of a decision tree. An increase in biomass may represent a transient phase of high productivity attributable to high growth and recruitment of small fish as a result of the removal of predators by fishing (e.g. in the North Sea) or environmental change (e.g. northern Humboldt and Portugal; Shannon *et al*., 2010). This emphasizes the need for a suite of ecological indicators to be derived that impart information about different properties of the ecosystem.

Perhaps the most problematic indicator is inverse fishing pressure, which resulted in significant trends in more than half the ecosystems. However, the estimation of both catch and biomass has challenges, ranging from bias and misreporting of catch data (Patterson, 1998; Watson and Pauly, 2001) to inconsistent survey protocols between years or across different fisheries within an ecosystem. It was assumed here that surveys were consistent through time and that catch was accurately reported, with no illegal, unreported, and unregulated (IUU) catches, although in some jurisdictions, this component may be considerable (Agnew *et al*., 2009). In addition, catch may be influenced by market pressures or management measures. Moreover, Blanchard *et al*. (2010) showed that inverse fishing pressure was significantly related to biomass in 10 of the 19 ecosystems. For these reasons, inverse fishing pressure was placed at the bottom of the decision tree. Auxiliary information in the form of additional indicators, such as total landings, total landings at a given trophic level, or information about management actions and policy could help elucidate the meaning of inverse fishing pressure.

For the 19 ecosystems examined here, inverse fishing pressure only determined the outcome of the decision tree for one ecosystem, Guinea (Decision Rule 1), despite three positive trends. Inverse fishing pressure decreased during the first 10 years of the dataseries, then remained relatively flat. This was reflected in the diagnosis for the period 1996–2005, which classified Guinea as not improving rather than deteriorating. Changes in the trophic level of landings and inverse fishing pressure suggest that, since 1985, the Guinean fishery was targeting higher trophic level species at a higher level of fishing mortality. This reflects the fishing history of this ecosystem: it has undergone rapid escalation of fishing effort and focused on coastal demersal species over the past two decades, accounting for the increase in the trophic level of the catch and the decrease in inverse fishing pressure (Lobry *et al*., 2003; Gascuel *et al*., 2009). However, exploitation in the offshore area has been more intense over a longer period (Laurans *et al*., 2004), and, as with other west African countries, IUU catches may be considerable (MRAG, 2005). Historically, exploitation in Guinea has been less intense than in Senegal or Mauritania (Lobry *et al*., 2003; Domalain *et al*., 2004).

Model structure

The decision-tree approach to classifying ecosystems into three states, improving, stationary (not improving), or deteriorating, is attractive for its simplicity and hence more likely to be acceptable to stakeholders. However, there are several methodological issues involved in this approach that are subjective, including the assessment of initial state (see above), the selection of ecosystem indicators, and the decision model itself, including the order of the indicators and the choice of decision rule.

All model outputs are sensitive to the design of the underlying model (Hilborn and Walters, 1992). Here, we explore the robustness of the final classification of the ecosystems to modifications in the order of indicators, and changes in the decision rules. Under the decision rules we used, the order of the indicators does not affect the final classification. However, if the decision rules were to be changed, the order of the indicators could have a larger impact on the outcome. For example, if an improving state was redefined as occurring when two positive trends were encountered, at which point the procedure was stopped, then Guinea would be classified as improving. If the order of the indicators were also changed, the Irish Sea, the North Sea, the Northeast United States, the northern Humboldt, and the southern Benguela could all be classified as improving, depending on the order. Clearly, it is important to have a decision rule that is robust to the order of the indicators and to have a clear rationale for the order in which the indicators are used. In practice, there needs to be buy-in to this reasoning and ordering by scientists and other stakeholders alike, if ecosystem classifications are to be accepted widely and acted upon by fishery managers. A logical next step would be to develop the decision-tree approach into an expert systems or computer-based decision-support systems approach. The latter has been shown to aid in the communication process with stakeholders and to integrate effectively the different types of data (Jarre *et al*., 2008).

The criteria for establishing decision rules

Decision rules remove many of the confounding issues that can occur when interpreting more than one indicator to assess the status of an ecosystem. However, care has to be taken in the development of decision rules, a process which should be undertaken in an interactive manner with scientists, managers, and stakeholders participating in setting the rules (FAO, 2003; Degnbol and Jarre, 2004). Explicit consideration of risk is an essential component of this process. Here, two risk-averse decision rules were explored, where a negative decrease in one or two ecosystem indicators was sufficient to classify an ecosystem as deteriorating. These were unbalanced rules because the criteria to classify an ecosystem as improving were more stringent. Relaxing these rules, or making them less risk-averse, would alter the classification of some of the ecosystems in Figure 1. At best, more ecosystems would be classified as not improving. For example, if a deteriorating ecosystem was defined by three negative trends, then of the 14 ecosystems classified as deteriorating under Decision Rule 1, only the North-central Adriatic Sea and the Northeast United States would remain in the category. The rest would be classified as not improving, which overall is not informative. A conservative decision rule is in accordance with the precautionary approach (FAO, 1995), where the risk of error is placed on the side of caution to avoid unacceptable or undesirable situations.

Conclusions

The strong take-home message from this exploratory decision-tree analysis is that many of the ecosystems examined here have been negatively impacted by fishing, as measured by the suite of ecosystem indicators, and are in the danger zone. They have been diagnosed to be in an ugly, deteriorating condition, having already been defined as impacted by the effects of fishing at the start of the time-series. Although it is likely that these ecosystems are being managed with a long-term goal of sustainability, the ecosystem attributes defined in Table 2 are endangered, as demonstrated by negative trends in the indicators. Here, only ecological indicators of the pressure and impacts of fishing were explored, but managing for sustainability requires consideration of biology, ecology, environment, economics, social aspects, and governance issues beyond simple stock dynamics (FAO, 2003; Bundy *et al*., 2008; Rice, 2008). Moreover, the potential effects of the environment, particularly in upwelling ecosystems (Shannon *et al*., 2010), need to be considered when interpreting such ecosystem indicators, and the decision model would need to be revised with improved knowledge. Our findings are supported by related studies: Link *et al*. (2010) explored drivers of ecosystem change, and Coll *et al*. (2010) used biotic factors to rank ecosystems and biotic, abiotic, and socio-economic factors to explain these rankings. The results of all these studies point to the same conclusion: for most of these ecosystems, we are not succeeding in managing, on an ecosystem basis, for long-term sustainability.

Acknowledgements

We thank all participants of the EurOceans IndiSeas Working Group (www.indiseas.org) for their inputs, ideas, and access to data: Pierre-François Baisnée (IRD, France), Nicolas Bez (IRD, France), Julia Blanchard (Cefas, UK), Maria de Fatima Borges (IPIMAR, Portugal), Pascal Cauquil (IRD, France), John Cotter (formerly Cefas, UK), Philippe Cury (IRD, France), Ibrahima Diallo (CNSHB, Guinea), Erich Diaz (IMARPE, Peru), Beth Fulton (CSIRO, Australia), Sheila Heymans (SAMS, Scotland), Edda Johannesen (IMR, Norway), Didier Jouffre (IRD, Senegal), Souad Kifani (INRH, Morocco), Pierre Labrosse (IMROP, Mauritania), Jason Link (NOAA, USA), Pierre Lopez (IRD, France), Steve Mackinson (Cefas, UK), Hicham Masski (INRH, Morocco), Kathrine Michalsen (IMR, Norway), Christian Möllmann (University of Hamburg, Germany), Henn Ojaveer (EMI, Estonia), Khairdine ould Mohamed Abdallahi (IMROP, Mauritania), Ian Perry (DFO, Canada), Jake Rice (DFO, Canada), Djiga Thiao (CRODT, Senegal), and Dawit Yemane (MCM, South Africa). Additionally, we thank Arnaud Bertrand

(IRD) for interpretation of the indicators for Peru, and AB thanks Mike Dowd (Dalhousie University) for statistical support, the DFO's Ecosystem Research Initiative for funding support and two internal referees, and most importantly, Astrid Jarre and Simon Greenstreet for their considered and extremely useful reviews of the draft manuscript. The IndiSeas WG was funded by the European Network of Excellence EUR-OCEANS (FP6, contract 511106), the European collaborative project MEECE (Marine Ecosystem Evolution in a Changing Environment; FP7, contract 212085), and IRD (Institut de Recherche pour le Développement). LJS was financially supported by the MEECE project and Astrid Jarre's South African Research Chair in Marine Ecology and Fisheries.

References

Agnew, D. J., Pearce, J., Pramod, G., Peatman, T., Watson, R., Beddington, J. R., and Pitcher, T. J. 2009. Estimating the worldwide extent of illegal fishing. PLoS One, 4(2). http://www.ncbi.nlm.nih.gov/pmc/articles/PMC2646833/.

Anon. 1991. Reports of the ICES Advisory Committee on Fishery Management, 1990. ICES Cooperative Research Report, 173.

Arancibia, H., and Neira, S. 2005. Long-term changes in the mean trophic level of central Chile fishery landings. Scientia Marina, 69: 295–300.

Argüelles, J., Tafur, R., Taipe, A., Villegas, P., Keyl, F., Dominguez, N., and Salazara, M. 2008. Size increment of jumbo flying squid *Dosidicus gigas* mature females in Peruvian waters, 1989–2004. Progress in Oceanography, 79: 308–312.

Arneri, E. 1996. Fisheries resources assessment and management in Adriatic and Ionian Seas. FAO Fisheries Report, 533: 7–20.

Auster, P. J., and Langton, R. W. 1999. The effects of fishing on fish habitat. *In* Fish Habitat: Essential Fish Habitat and Rehabilitation, pp. 150–187. Ed. by L. Benaka. American Fisheries Society Symposium, 22. 400 pp.

Bager, M., Søndergaard, M. K., and MacKenzie, B. R. 2007. The development of fisheries at Bornholm, Denmark (Baltic Sea) during 1880s–1914. Fisheries Research, 87: 146–154.

Bagge, O., Thurow, F., Steffensen, E., and Bay, J. 1994. The Baltic cod. Dana, 10: 1–28.

Bakkala, R. G. 1993. Structure and historical changes in the groundfish complex of the eastern Bering Sea. NOAA Technical Report, NMFS 114. 91 pp. (http://spo.nwr.noaa.gov/tr114opt.pdf)

Ball, B. J., Fox, G., and Munday, B. W. 2000. Long- and short-term consequences of a *Nephrops* trawl fishery on the benthos and environment of the Irish Sea. ICES Journal of Marine Science, 57: 1315–1320.

Ballón, M., Wosnitza-Mendo, C., Guevara-Carrasco, R., and Bertrand, A. 2008. The impact of overfishing and *El Niño* on the condition factor and reproductive success of Peruvian hake, *Merluccius gayi peruanus*. Progress in Oceanography, 79: 300–307.

Bas, C., Morales, E., and Rubio, M. 1955. La pesca en España. 1. Cataluña. Instituto de Investigaciones Pesqueras. 468 pp.

Belton, V., and Stewart, T. J. 2002. Multiple Criteria Decision Analysis: an Integrated Approach, 2nd printing. Kluwer Academic, Dordrecht. 370 pp.

Bender, K. M. 2007. Global fish production and climate change. Proceedings of the National Academy of Sciences of the USA, 104: 19701–19714.

Bertrand, A., Segura, M., Gutiérrez, M., and Vásquez, L. 2004. From small-scale habitat loopholes to decadal cycles: a habitat-based hypothesis explaining fluctuation in pelagic fish populations off Peru. Fish and Fisheries, 5: 296–316.

Bertrand, S., Díaz, E., and Lengaigne, M. 2008. Patterns in the spatial distribution of Peruvian anchovy (*Engraulis ringens*) revealed by spatially explicit fishing data. Progress in Oceanography, 79: 379–389.

Best, P. B., Crawford, R. J. M., and van der Elst, R. P. 1997. Top predators in Southern Africa's marine ecosystems. Transactions of the Royal Society of South Africa, 52: 177–225.

Blanchard, J. L., Coll, M., Trenkel, V. M., Vergnon, R., Yemane, D., Jouffre, D., Link, J. S., *et al.* 2010. Trend analysis of indicators: a comparison of recent changes in the status of marine ecosystems around the world. ICES Journal of Marine Science, 67: 732–744.

Bradshaw, C., Veale, L. O., and Brand, A. R. 2002. The role of scallop-dredge disturbance in long-term changes in Irish Sea benthic communities: a re-analysis of an historical dataset. Journal of Sea Research, 47: 161–184.

Brander, K. 1980. Fisheries management and conservation in the Irish Sea. Helgoland Marine Research, 33: 687–699.

Brander, K. 1981. Disappearance of common skate *Raia batis* from Irish Sea. Nature, 290: 48–49.

Breeze, H. 2002. Commercial fisheries of the Sable Gully and surrounding region: historical and present activities. Canadian Manuscript Report of Fisheries and Aquatic Sciences, 2612. viii+83 pp.

Bundy, A. 2005. Structure and function of the eastern Scotian Shelf ecosystem before and after the groundfish collapse in the early 1990s. Canadian Journal of Fisheries and Aquatic Sciences, 62: 1453–1473.

Bundy, A., Chuenpagdee, R., Jentoft, S., and Mahon, R. 2008. If science is not the answer, what is? An alternative governance model for reversing the dismal state of the world's fisheries resources. Frontiers in Ecology and the Environment, 6: 152–155.

Caddy, J. F. 2002. Limit reference points, traffic lights, and holistic approaches to fisheries management with minimal stock assessment input. Fisheries Research, 56: 133–137.

Caddy, J. F. 2004. Current usage of fisheries indicators and reference points, and their potential application to management of fisheries for marine invertebrates. Canadian Journal of Fisheries and Aquatic Sciences, 61: 1307–1324.

Caddy, J. F., Csirke, J., Garcia, S. M., and Grainger, R. J. R. 1998. How pervasive is "fishing down marine food webs"? Science, 282: 1383a.

Castillo, S., and Mendo, J. 1987. Estimation of unregistered Peruvian anchoveta (*Engraulis ringens*) in official catch statistics, 1951–1982. *In* The Peruvian Anchoveta and its Upwelling Ecosystem: Three Decades of Change, pp. 109–116. Ed. by D. Pauly, and I. Tsukayama. ICLARM Studies and Reviews, 15. 351 pp.

Cazelles, B. 2004. Symbolic dynamics for identifying similarity between rhythms of ecological time series. Ecology Letters, 7: 755–763.

Chavance, P., Bâ, M., Gascuel, D., Vakily, J. M., and Pauly, D. (Eds). 2004. Pêcheries maritimes, écosystèmes et sociétés en Afrique de l'Ouest: un demi-siècle de changement [Marine fisheries, ecosystems and societies in West Africa: half a century of change]. Actes du Symposium International, Dakar (Sénégal), 24–28 Juin 2002. Office des Publications Officielles des Communautés Européennes, Bruxelles. Collections des Rapports de Recherche Halieutique ACP-UE, 15. xxxvi+532 pp. +xiv.

Chavez, F. P., Bertrand, A., Guevara-Carrasco, R., Soler, P., and Csirke, J. 2008. The northern Humboldt Current system: brief history, present status and a view towards the future. Progress in Oceanography, 79: 95–105.

Chavez, F. P., Ryan, J., Lluch-Cota, S. E., and Ñiquen, M. 2003. From anchovies to sardines and back: multidecadal change in the Pacific Ocean. Science, 299: 217–221.

Cheung, W. W. L., Lam, V. W. Y., Sarmiento, J. L., Kearney, K., Watson, R., and Pauly, D. 2009. Projecting global marine biodiversity impacts under climate change scenarios. Fish and Fisheries, 10: 235–251.

Cochrane, K. L., Augustyn, C. J., Cockcroft, A. C., David, J. H. M., Griffiths, M. H., Groeneveld, J. C., Lipiński, M. R., *et al.* 2004.

An ecosystem approach to fisheries in the southern Benguela context. African Journal of Marine Science, 26: 9–35.

Coll, M., Libralato, S., Tudela, S., Palomera, I., and Pranovi, F. 2008a. Ecosystem overfishing in the ocean. PLoS One, 3: e3881.

Coll, M., Palomera, I., Tudela, S., and Dowd, M. 2008b. Food-web dynamics in the South Catalan Sea ecosystem (NW Mediterranean) for 1978–2003. Ecological Modelling, 217: 95–116.

Coll, M., Santojanni, A., Arneri, E., Palomera, I., and Tudela, S. 2007. An ecosystem model of the Northern and Central Adriatic Sea: analysis of ecosystem structure and fishing impacts. Journal of Marine Systems, 67: 119–154.

Coll, M., Santojanni, A., Palomera, I., and Arneri, E. 2009. Food-web dynamics in the north-central Adriatic marine ecosystem (Mediterranean Sea) over the last three decades. Marine Ecology Progress Series, 381: 17–37.

Coll, M., Shannon, L. J., Yemane, D., Link, J. S., Ojaveer, H., Neira, S., Jouffre, D., *et al.* 2010. Ranking the ecological relative status of exploited marine ecosystems. ICES Journal of Marine Science, 67: 769–786.

Crawford, R. J. M. 1999. Seabird responses to long-term changes of prey resources off southern Africa. *In* Proceedings of the 22nd International Ornithological Congress, pp. 688–705. Ed. by N. J. Adams, and R. H. Slotow. BirdLife South Africa, Johannesburg.

Crawford, R. J. M., and Dyer, B. M. 1995. Responses by four seabird species to a fluctuating availability of Cape anchovy *Engraulis capensis* off South Africa. Ibis, 137: 329–339.

Crawford, R. J. M., and Jahncke, J. 1999. Comparison of trends in abundance of guano-producing seabirds in Peru and southern Africa. South African Journal of Marine Science, 21: 145–156.

Cury, P. M., Mullon, C., Garcia, S. M., and Shannon, L. J. 2005b. Viability theory for an ecosystem approach to fisheries. ICES Journal of Marine Science, 62: 577–584.

Cury, P. M., Shannon, L. J., Roux, J-P., Daskalov, G. M., Jarre, A., Moloney, C. L., and Pauly, D. 2005a. Trophodynamic indicators for an ecosystem approach to fisheries. ICES Journal of Marine Science, 62: 430–442.

Daan, N., Christensen, V., and Cury, P. (Eds). 2005. Quantitative ecosystem indicators for fisheries management. ICES Journal of Marine Science, 62: 307–614.

Dardignac, J. 1988. Les pêcheries du Golfe de Gascogne. Bilan des connaissances. Rapports scientifiques et techniques de l'IFREMER, 9. 204 pp.

Degnbol, P. 2005. Indicators as a means of communicating knowledge. ICES Journal of Marine Science, 62: 606–611.

Degnbol, P., and Jarre, A. 2004. Review of indicators in fisheries management—a development perspective. African Journal of Marine Science, 26: 303–326.

DFO (Department of Fisheries and Oceans). 2003. State of the eastern Scotian Shelf ecosystem. Canadian Science Advisory Secretariat Ecosystem Status Report, 003/004.

Domain, F. 1980. Contribution à la connaissance de l'écologie des poissons démersaux du plateau continental sénégalo-mauritanien: les ressources démersales dans le contexte général du golfe de Guinée. Thèse de doctorat ès - Sciences Naturelles de l'Université Pierre et Marie Curie – Paris VI, Volume 1.

Domain, F. 1999. Influence de la pêche et de l'hydro-climat sur l'évolution dans le temps du stock côtier (1985–1995). *In* La Pêche Côtière en Guinée: Ressources et Exploitation, pp. 7–27. Ed. by F. Domain, P. Chavance, and A. Diallo. IRD édition, Paris, Collection Colloques et Séminaire, France. 393 pp.

Domalain, G., Jouffre, D., Thiam, D., Traoré, S., and Wang, L. 2004. Évolution de la diversité spécifique dans les campagnes de chalutage démersales du Sénégal et de la Guinée. *In* Pêcheries Maritimes, Écosystèmes et Sociétés en Afrique de l'Ouest: un Demi-siècle de Changement [Marine Fisheries, Ecosystems and Societies in West Africa: Half a Century of Change], pp. 299–310. Ed. by P. Chavance, M. Bâ, D. Gascuel, J. M. Vakily, and D. Pauly. Actes du Symposium International, Dakar (Sénégal), 24–28 Juin 2002. Office des Publications Officielles des Communautés Européennes, Bruxelles. Collections des Rapports de Recherche Halieutique ACP-UE, 15. xxxvi+532 pp.+xiv.

Duplisea, D. E., Kerr, S. R., and Dickie, L. M. 1997. Demersal fish biomass size spectra on the Scotian Shelf, Canada: species replacement at the shelfwide scale. Canadian Journal of Fisheries and Aquatic Sciences, 54: 1725–1735.

Ecosystem Assessment Program. 2009. Ecosystem Status Report For the Northeast U.S. Continental Shelf Large Marine Ecosystem. Northeast Fisheries Science Center Reference Document, 09-11. 34 pp.

Eero, M., MacKenzie, B. R., Karlsdottir, H. M., and Gaumiga, R. 2007. Development of international fisheries for the eastern Baltic cod (*Gadus morhua*) from the late 1880s until 1938. Fisheries Research, 87: 155–166.

Essington, T. E., Beaudreau, A. H., and Wiedenmann, J. 2006. Fishing through marine food webs. Proceedings of the National Academy of Sciences of the USA, 103: 3171–3175.

FAO. 1995. Precautionary approach to fisheries. 1. Guidelines on the precautionary approach to capture fisheries and species introductions. Elaborated by the Technical Consultation on the Precautionary Approach to Capture Fisheries (including Species introductions), Lysekil, Sweden, 6–13 June 1995. FAO Fisheries Technical Paper, 350, Part 1. 52 pp.

FAO. 2003. Fisheries Management. 2. The Ecosystem Approach to Fisheries. FAO Technical Guidelines for Responsible Fisheries, 4(Suppl. 2). 112 pp.

FAO. 2006. Rapport for the FAO/CECAF Working Group on the Assessment of Demersal Ressources Subgroup, Saly, Senegal, 14–23 September 2004. CECAF/ECAF Series, 06/68. 54 pp.

Farrugio, H., Oliver, P., and Biagi, F. 1993. An overview of the history, knowledge, recent and future research trends in the Mediterranean fisheries. Scientia Marina, 57: 105–119.

Fontana, A., and Weber, J. 1982. Aperçu de la situation de la pêche maritime sénégalaise (Déc -82). Document scientifique du Centre Océanographique de Dakar Thiaroye, ISRA, Dakar (Senegal). 34 pp.

Forest, A. 2001. Ressources halieutiques hors quotas du Nord Est Atlantique: bilan des connaissances et analyse de scénarios d'évolution de la gestion. Contrat Ifremer/MAPA, Réf. 99-l1-03-01. 390 pp.

Garcia, S. M., Zerbi, A., Aliaume, C., Do Chi, T., and Lasserre, G. 2003. The ecosystem approach to fisheries: issues, terminology, principles, institutional foundations, implementation and outlook. FAO Fisheries Technical Paper, 443. 71 pp.

Gascuel, D., Guénette, S., Diallo, I., and Sidibé, A. 2009. Impact de la pêche sur l'écosystème marin de Guinée - modélisation EwE 1985/2005 (Fishing impact on the Guinean marine ecosystem: a 1985/2005 model using EwE). Fisheries Centre Research Report, 17(4). 60 pp.

Gascuel, D., Labrosse, P., Meissa, B., Taleib Sidi, M. O., and Guénette, S. 2007. The decline of demersal resources in North-West Africa: an analysis of Mauritanian trawl survey data over the last 25 years. African Journal of Marine Science, 29: 331–345.

Gascuel, D., Laurans, M., Sidibé, A., and Barry, M. D. 2004. Diagnostic comparatif de l'état des stocks et évolutions d'abondance des ressources démersales dans les pays de la CSRP. *In* Pêcheries Maritimes, Écosystèmes et Sociétés en Afrique de l'Ouest: un Demi Siècle de Changement, [Marine Fisheries, Ecosystems and Societies in West Africa: Half a Century of Change], pp. 205–222. Ed. by P. Chavance, M. Bâ, D. Gascuel, J. M. Vakily, and D. Pauly. Actes du Symposium International, Dakar (Sénégal), 24–28 Juin 2002. Office des Publications Officielles des Communautés Européennes, Bruxelles. Collections des Rapports de Recherche Halieutique ACP-UE, 15. xxxvi+532 pp. +xiv.

Gjøsæter, H. 1995. Pelagic fish and the ecological impact of the modern fishing industry in the Barents Sea. Arctic, 48: 267–278.

Grainger, R. J. R., and Garcia, S. M. 1996. Chronicles of marine fishery landings (1950–1994). Trend analysis and fisheries potential. FAO Fisheries Technical Paper, 359. 51 pp.

Griffiths, M. H. 2000. Long-term trends in catch and effort of commercial linefish off South Africa's Cape Province: snapshots of the 20th century. South African Journal of Marine Science, 22: 81–110.

Guénette, S., and Diallo, I. 2004. Exploration d'un modèle préliminaire de l'écosystème marin de Guinée. *In* Pêcheries Maritimes, Écosystèmes et Sociétés en Afrique de l'Ouest: un Demi Siècle de Changement, [Marine Fisheries, Ecosystems and Societies in West Africa: Half a Century of Change], pp. 329–346. Ed. by P. Chavance, M. Bâ, D. Gascuel, J. M. Vakily, and D. Pauly. Actes du Symposium International, Dakar (Sénégal), 24–28 Juin 2002. Office des Publications Officielles des Communautés Européennes, Bruxelles. Collections des Rapports de Recherche Halieutique ACP-UE, 15. xxxvi+532 pp. +xiv.

Guevara-Carrasco, R., and Lleonart, J. 2008. Dynamics and fishery of the Peruvian hake: between the nature and the man. Journal of Marine Systems, 71: 249–259.

Gutiérrez, D., Sifeddine, A., Field, D. B., Ortlieb, L., Vargas, G., Chávez, F., Velazco, F., *et al.* 2008. Rapid reorganization in ocean biogeochemistry off Peru towards the end of the Little Ice Age. Biogeosciences Discussions, 5: 3919–3943.

Halliday, R. G., Fanning, L. P., and Mohn, R. K. 2001. Use of the traffic light method in fishery management planning. Canadian Stock Assessment Secretariat Research Document, 2001/18. 41 pp.

Halliday, R. G., and Pinhorn, A. T. 1996. North Atlantic fishery management systems: a comparison of management methods and resource trends. Journal of Northwest Atlantic Fishery Science, 20. 140 pp.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., *et al.* 2008. A global map of human impact on marine ecosystems. Science, 319: 948–952.

Hamre, J. 1994. Biodiversity and exploitation of the main fish stocks in the Norwegian—Barents Sea ecosystem. Biodiversity and Conservation, 3: 473–492.

Hays, G. C., Richardson, A. J., and Robinson, C. 2005. Climate change and marine plankton. Trends in Ecology and Evolution, 20: 337–344.

Hilborn, R., and Walters, C. J. 1992. Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty. Chapman and Hall, New York. 570 pp.

Hill, A. S., Veale, L. O., Pennington, D., Whyte, S. G., Brand, A. R., and Hartnoll, R. G. 1999. Changes in Irish Sea benthos: possible effects of 40 years of dredging. Estuarine, Coastal and Shelf Science, 48: 739–750.

Hoag, S. 1971. Effects of domestic trawling on the halibut stocks of British Columbia. INPHC Scientific Report, 53. 18 pp.

Hollingworth, C. (Ed). 2000. Ecosystem effects of fishing. Proceedings of an ICES/SCOR Symposium held in Montpellier, France, 16–19 March 1999. ICES Journal of Marine Science, 57: 465–792.

Hooper, D. U., Chapin, F. S., Ewel, J. J., Hector, A., Inchausti, P., Lavorel, S., Lawton, H., *et al.* 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecological Monographs, 75: 3–35.

ICES. 1981. Report of Mackerel Working Group. ICES Document CM 1981/H: 7. 73 pp.

ICES. 2007. Report of the Working Group on the Assessment of Southern Shelf Stocks of Hake, Monk and Megrim (WGHMM), 8–17 May 2007, Vigo, Spain. ICES Document CM 2007/ACFM: 21. 700 pp.

ICES. 2008a. The effect of climate change on the distribution and abundance of marine species in the OSPAR Maritime Area. ICES Cooperative Research Report, 293. 293 pp.

ICES. 2008b. Report of the Baltic Fisheries Assessment Working Group (WGBFAS), 8–17 April 2008, ICES Headquarters, Copenhagen. ICES Document CM 2008/ACOM: 06. 692 pp.

IMROP. 2007. Rapport de synthèse du 6ème Groupe de Travail sur l'évaluation des ressources et l'aménagement des pêcheries de la ZEE mauritanienne. Document IMROP. 26 pp.

Jackson, J. B. C., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., Bradbury, *et al.* 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science, 293: 629–638.

Jarre, A., Moloney, C. L., Shannon, L. J., Fréon, P., van der Lingen, C. D., Verheye, H., Hutchings, L., *et al.* 2006. Detecting and forecasting long-term ecosystem changes. *In* Benguela: Predicting a Large Marine Ecosystem, pp. 239–272. Ed. by L. V. Shannon, G. Hempel, P. Malanotte-Rizzoli, C. L. Moloney, and J. Woods. Large Marine Ecosystem Series, 14. Elsevier, Amsterdam. 410 pp.

Jarre, A., Paterson, B., Moloney, C. L., Miller, D. G. M., Field, J. G., and Starfield, A. 2008. Knowledge-based systems as decision support tools in an ecosystem approach to fisheries: comparing a fuzzy-logic and a rule-based approach. Progress in Oceanography, 79: 390–400.

Jennings, S. 2005. Indicators to support an ecosystem approach to fisheries. Fish and Fisheries, 6: 212–232.

Jennings, S., and Blanchard, J. L. 2004. Fish abundance with no fishing: predictions based on macroecological theory. Journal of Animal Ecology, 73: 632–642.

Jennings, S., and Dulvy, N. K. 2005. Reference points and reference directions for size-based indicators of community structure. ICES Journal of Marine Science, 62: 397–404.

Jouffre, D., Borges, M. F., Bundy, A., Coll, M., Diallo, I., Fulton, E. A., Guitton, J., *et al.* 2010. Estimating EAF indicators from scientific trawl surveys: theoretical and practical concerns. ICES Journal of Marine Science, 67: 796–806.

Jukic-Peladica, S., Vrgoc, N., Krstulovic-Sifner, S., Piccinetti, C., Piccinetti-Manfrin, G., Marano, G., and Ungaro, N. 2001. Long-term changes in demersal resources of the Adriatic Sea: comparison between trawl surveys carried out in 1948 and 1998. Fisheries Research, 53: 95–104.

Kaiser, M., and DeGroot, S. J. (Eds). 2000. Effects of Fishing on Non-target Species and Habitats Biological, Conservation and Socio-economic Issues. Blackwell, Oxford, UK. 416 pp.

Kaiser, M. J., Hill, A. S., Ramsay, K., Spencer, B. E., Brand, A. R., Veale, L. O., Prudden, K., *et al.* 1996. Benthic disturbance by fishing gear in the Irish Sea: a comparison of beam trawling and scallop dredging. Aquatic Conservation: Marine and Freshwater Ecosystems, 6: 269–285.

Keeney, R. L., and Raiffa, H. 1976. Decisions with Multiple Objectives: Preferences and Value Tradeoffs. John Wiley, New York. 596 pp.

Kifani, S., Masskia, H., and Faraj, A. 2008. The need of an ecosystem approach to fisheries: the Moroccan upwelling-related resources case. Fisheries Research, 94: 36–42.

Koeller, P., Savard, L., Parsons, D. G., and Fu, C. 2000. A precautionary approach to assessment and management of shrimp stocks in the Northwest Atlantic. Journal of Northwest Atlantic Fishery Science, 27: 235–246.

Krost, P. 1990. Der Einfluss der Grundschleppnetzfischerei auf Nahrsalz-Freisetzung aus dem Sediment und Makrofauna der Kieler Bucht (Westl. Ostsee). Berichte aus dem Institut für Meereskunde an der Christian-Albrects-Universität Kiel, 200.

Krost, P., Bernhard, M., Werner, F., and Hukriede, W. 1990. Otter trawl tracks in Kiel Bay (western Baltic) mapped by side-sonar. Meeresforschungen, 32: 344–353.

Laurans, M., Gascuel, D., Chassot, E., and Thiam, D. 2004. Changes in the trophic structure of fish demersal communities in West Africa in the three last decades. Aquatic Living Resources, 17: 163–173.

Lindeboom, H. J., and de Groot, S. J. 1998. IMPACT-II: the effects of different types of fisheries on the North Sea and Irish Sea benthic ecosystems. NIOZ-rapport, 1998(1). Netherlands Institute for Sea Research, Den Burg, Texel, The Netherlands. 404 pp.

Link, J. S. 2005. Translating ecosystem indicators into decision criteria. ICES Journal of Marine Science, 62: 569–576.

Link, J. S. 2007. Under-appreciated species in ecology: "ugly fish" in the Northwest Atlantic Ocean. Ecological Applications, 17: 2037–2060.

Link, J. S., and Brodziak, J. K. T. (Eds). 2002. Report on the Status of the NE US Continental Shelf Ecosystem. NEFSC Ecosystem Status Working Group. Northeast Fisheries Science Center Reference Document, 02-11. 245 pp.

Link, J. S., Brodziak, J. K. T., Edwards, S. F., Overholtz, W. J., Mountain, D., Jossi, J. W., Smith, T. D., *et al.* 2002. Marine ecosystem assessment in a fisheries management context. Canadian Journal of Fisheries and Aquatic Sciences, 59: 1429–1440.

Link, J. S., Griswold, C. A., Methratta, E. M., and Gunnard, J. (Eds). 2006. Documentation for the Energy Modeling and Analysis eXercise (EMAX). Northeast Fisheries Science Center Reference Document, 06-15. 166 pp.

Link, J. S., Yemane, D., Shannon, L. J., Coll, M., Shin, Y-J., Hill, L., and Borges, M. F. 2010. Relating marine ecosystem indicators to fishing and environmental drivers: an elucidation of contrasting responses. ICES Journal of Marine Science, 67: 787–795.

Lobry, J., Gascuel, D., and Domain, F. 2003. La biodiversité spécifique des ressources démersales du plateau continental guinéen: utilisation d'indices classiques pour le diagnostic sur l'évolution de l'écosystème. Aquatic Living Resources, 16: 59–68.

Longhurst, A. 2002. Murphy's law revisited: longevity as a factor in recruitment to fish populations. Fisheries Research, 56: 125–131.

Lostado, R. B., Del Río, V. O., Vivas, A. D., Alegret, J. L. l., Franquesa, R., Lleonart, J., Oliver, P., *et al.* 1999. The Castelló Trawling Project (1961–1966): teachings for a sustainable fisheries management. FAO–COPEMED. 173 pp.

Lotze, H. K., Lenihan, H. S., Bourque, B. J., Bradbury, R. H., Cooke, R. G., Kay, M. C., Kidwell, S. M., *et al.* 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science, 312: 1806–1809.

Lotze, H. K., and Milewski, I. 2004. Two centuries of multiple human impacts and successive changes in a North Atlantic food web. Ecological Applications, 14: 1428–1447.

Mackinson, S., and Daskalov, G. 2007. An ecosystem model of the North Sea to support an ecosystem approach to fisheries management: description and parameterisation. Science Series Technical Report, Cefas Lowestoft, 142. 196 pp.

Mardle, S., and Pascoe, S. 1999. A review of applications of multiple criteria decision-making techniques to fisheries. Marine Resource Economics, 14: 41–63.

Matishov, G. G., Denisov, V. V., Dzhenyuk, S. L., Karamushko, O. V., and Daler, D. 2004. The impact of fisheries on the dynamics of commercial fish species in Barents Sea and the Sea of Azov, Russia: a historical perspective. Ambio, 33: 63–67.

McDaniels, T. L. 1995. Using judgment in resource management: a multiple objective analysis of a fisheries management decision. Operations Research, 43: 415–426.

Methratta, E. T., and Link, J. S. 2006. Evaluation of quantitative indicators for marine fish communities. Ecological Indicators, 6: 575–588.

MRAG. 2005. Review of impacts of Illegal, Unreported and Unregulated fishing on developing countries. MRAG, London. http://www.dfid.gov.uk/pubs/files/illegal-fishing-mrag-report.pdf.

Neira, S. 2008. Assessing the effects of internal (trophic structure) and external (fishing and environment) forcing factors on fisheries off central Chile: basis for an ecosystem approach to management. PhD thesis, University of Cape Town. 254 pp. +2 Appendices.

Nicholson, M. D., and Jennings, S. 2004. Testing candidate indicators to support ecosystem-based management: the power of monitoring surveys to detect temporal trends in fish community metrics. ICES Journal of Marine Science, 61: 35–42.

NRC. 1996. The Bering Sea Ecosystem. Committee on the Bering Sea Ecosystem, National Academy of Sciences. 308 pp. http://www.nap.edu/catalog.php?record_id=5039.

Olsen, N., Boutillier, J. A., and Convey, L. 2000. Estimated bycatch in the British Columbia shrimp trawl fishery. Canadian Stock Assessment Secretariat Research Document, 2000/168. 54 pp.

Palomera, I., Olivar, M. P., Salat, J., Sabatés, A., Coll, M., García, A., and Morales-Nin, B. 2007. Small pelagic fish in the NW Mediterranean Sea: an ecological review. Progress in Oceanography, 74: 377–396.

Papaconstantinou, C., and Farrugio, H. 2000. Fisheries in the Mediterranean. Mediterranean Marine Science, 1: 5–18.

Paterson, B., Jarre, A., Moloney, C. L., Fairweather, T. P., van der Lingen, C. D., Shannon, L. J., and Field, J. G. 2007. A fuzzy-logic tool for multi-criteria decision making in fisheries: the case of the South African pelagic fishery. Marine and Freshwater Research, 58: 1056–1068.

Patterson, K. 1998. Assessing fish stocks when catches are misreported: model, simulation tests, and application to cod, haddock, and whiting in the ICES Area. ICES Journal of Marine Science, 55: 878–891.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., and Torres, F. J. 1998a. Fishing down marine food webs. Science, 279: 860–863.

Pauly, D., Froese, R., and Christensen, V. 1998b. How pervasive is fishing down marine the food webs? Response to Caddy *et al.* Science, 282: 1383a.

Pauly, D., Muck, P., Mendo, J., and Tsukayama, I. (Eds). 1989. The Peruvian upwelling ecosystem: dynamics and interactions. ICLARM Conference Proceedings, 18. 438 pp.

Pauly, D., and Palomares, M. L. 1989. New estimates of monthly biomass, recruitment and related statistics of anchoveta (*Engraulis ringens*) off Peru (4–14°S), 1953–1985. *In* The Peruvian Upwelling Ecosystem: Dynamics and Interactions, pp. 189–206. Ed. by D. Pauly, P. Muck, J. Mendo, and I. Tsukayama. ICLARM Conference Proceedings, 18. 438 pp.

Pauly, D., Palomares, M. L., Froese, R., Pascualita, S., Vakily, M., Preikshot, D., and Wallace, S. 2001. Fishing down Canadian aquatic food webs. Canadian Journal of Fisheries and Aquatic Sciences, 58: 51–62.

Pauly, D., Palomares, M. L., and Gayanilo, F. C. 1987. VPA estimates of monthly population length composition, recruitment, mortality, biomass and related statistics of Peruvian anchoveta, 1953 to 1981. *In* The Peruvian Anchoveta and its Upwelling Ecosystem: Three Decades of Change, pp. 167–178. Ed. by D. Pauly, and I. Tsukayama. ICLARM Studies and Reviews, 15. 351 pp.

Pauly, D., and Tsukayama, I. (Eds). 1987. The Peruvian anchoveta and its upwelling ecosystem: three decades of change. ICLARM Studies and Reviews, 15. 351 pp.

Pauly, D., Watson, R., and Adler, J. 2005. Global trends in world fisheries: impacts on marine ecosystems and food security. Philosophical Transactions of the Royal Society of London, Series B: Biological Sciences, 360: 5–12.

Pitcher, T. J., Kalikoski, D., Short, K., Varkey, D., and Pramoda, G. 2009. An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. Marine Policy, 33: 223–232.

Quéro, J-C., and Cendrero, O. 1996. Incidence de la pêche sur la biodiversité ichtyologique marine: le bassin d'Arcachon et le plateau continental Sud Gascogne. Cybium, 20: 232–356.

Rice, J. 2003. Environmental health indicators. Ocean and Coastal Management, 46: 235–259.

Rice, J. 2005. Ecosystems effects of fishing: impacts, metrics, and management strategies. ICES Cooperative Research Report, 272. 177 pp.

Rice, J. 2008. Can we manage ecosystems in a sustainable way? Journal of Sea Research, 60: 8–20.

Rochet, M-J., and Rice, J. C. 2005. A framework for selecting a suite of indicators for fisheries management. ICES Journal of Marine Science, 62: 516–527.

Rochet, M-J., and Trenkel, V. M. 2005. Which community indicators can measure the impact of fishing? A review and proposals. Canadian Journal of Fisheries and Aquatic Sciences, 60: 86–99.

Rochet, M-J., Trenkel, V. M., Bellail, R., Coppin, F., Le Pape, O., Mahé, J-C., Morin, J., *et al.* 2005. Combining indicator trends to assess ongoing changes in exploited fish communities: diagnostic of communities off the coasts of France. ICES Journal of Marine Science, 62: 1647–1664.

Rogers, S. I., and Greenaway, B. 2005. A UK perspective on the development of marine ecosystem indicators. Marine Pollution Bulletin, 50: 9–19.

Rosenberg, A. A., Bolster, W. J., Alexander, K. E., Leavenworth, W. B., Cooper, A. B., and McKenzie, M. G. 2005. The history of ocean resources: modeling cod biomass using historical records. Frontiers in Ecology and the Environment, 3: 78–84.

Rumohr, H., and Krost, P. 1991. Experimental evidence of damage to benthos by bottom trawling with special reference to *Arctica islandica*. Meeresforschungen, 33: 340–345.

Ryther, J. H. 1969. Photosynthesis and fish production in the sea. Science, 166: 72–76.

Sardà, F. 1998. Symptoms of overexploitation in the stock of the Norway lobster (*Nephrops norvegicus*) on the "Serola Bank" (western Mediterranean Sea off Barcelona). Scientia Marina, 62: 295–299.

Shannon, L. J., Coll, M., Yemane, D., Jouffre, D., Neira, S., Betrand, A., Diaz, E., *et al.* 2010. Comparing data-based indicators across upwelling and comparable systems for communicating ecosystem states and trends. ICES Journal of Marine Science, 67: 807–832.

Shannon, L. J., Neira, S., and Taylor, M. 2008. Comparing internal and external drivers in the southern Benguela and the southern and northern Humboldt upwelling ecosystems. African Journal of Marine Science, 30: 63–84.

Shin, Y-J., Rochet, M-J., Jennings, S., Field, J. G., and Gislason, H. 2005. Using size-based indicators to evaluate the ecosystem effects of fishing. ICES Journal of Marine Science, 62: 384–396.

Shin, Y-J., and Shannon, L. J. 2010. Using indicators for evaluating, comparing, and communicating the ecological status of exploited marine ecosystems. 1. The IndiSeas Project. ICES Journal of Marine Science, 67: 686–691.

Shin, Y-J., Shannon, L. J., Bundy, A., Coll, M., Aydin, K., Bez, N., Blanchard, J. L., *et al.* 2010. Using indicators for evaluating, comparing, and communicating the ecological status of exploited marine ecosystems. 2. Setting the scene. ICES Journal of Marine Science, 67: 692–716.

Sinclair, A. F., Swain, D. P., and Hanson, J. M. 2002. Measuring changes in the direction and magnitude of size-selective mortality in a commercial fish population. Canadian Journal of Fisheries and Aquatic Sciences, 59: 361–371.

Stempniewicz, L. 1994. Marine birds drowning in fishing nets in the Gulf of Gdansk (southern Baltic): numbers, species composition, age and sex structure. Ornis Svecica, 4: 123–132.

Stokes, T. K., McGlade, J. M., and Law, R. (Eds). 1993. The Exploitation of Evolving Resources. Lecture Notes in Biomathematics, 99. Springer, Berlin. 264 pp.

Taylor, M. H., Tam, J., Blaskovic, V., Espinoza, P., Ballón, R. M., Wosnitza-Mendo, C., Argüelles, J., *et al.* 2008. Trophic modeling of the northern Humboldt Current ecosystem. 2. Elucidating ecosystem dynamics from 1995–2004 with a focus on the impact of ENSO. Progress in Oceanography, 79: 366–378.

Tovar, H., Guillen, V., and Nakama, M. E. 1987. Monthly population size of three guano bird species off Peru, 1953 to 1982. *In* The Peruvian Anchoveta and its Upwelling Ecosystem: Three Decades of Change, pp. 208–218. Ed. by D. Pauly, and I. Tsukayama. ICLARM Studies and Reviews, 15. 351 pp.

Trenkel, V. M., Rochet, M-J., and Mesnil, B. 2007. From model-based prescriptive advice to indicator-based interactive advice. ICES Journal of Marine Science, 64: 768–774.

Vrgoc, N., Arneri, E., Jukic-Peladica, S., Krstulovic Šifner, S., Mannini, P., Marceta, B., Osmani, K., *et al.* 2004. Review of current knowledge on shared demersal stocks of the Adriatic Sea. GCP/RER/010/ITA/TD-12. AdriaMed Technical Documents, 12. 91 pp.

Waddell, B. J., and Ware, D. M. 1995. Historic domestic and foreign fish catch statistics for British Columbia: 1920–1991. Canadian Technical Report of Fisheries and Aquatic Sciences, 2030. 37 pp.

Waring, G. T., Pace, R. M., Quintal, J. M., Fairfield, C. P., and Maze-Foley, K. 2004. US Atlantic and Gulf of Mexico Marine Mammal Stock Assessments—2003. NOAA Technical Memorandum, NMFS-NE-182. www.nefsc.noaa.gov/nefsc/publications/tm/tm182.

Watson, R., and Pauly, D. 2001. Systematic distortions in world fisheries catch trends. Nature, 414: 534–536.

Yemane, D., Field, J. G., and Griffiths, M. H. 2004. Effects of fishing on the size and dominance structure of linefish of the Cape region, South Africa. African Journal of Marine Science, 26: 161–177.

Yemane, D., Field, J. G., and Leslie, R. W. 2010. Spatio-temporal patterns in the diversity of demersal fish communities off the south coast of South Africa. Marine Biology, 157: 269–281.

Appendix A

Results of the assessment of initial state of the 19 ecosystems evaluated. An ecosystem was defined as impacted where two of the following criteria applied at the beginning of the time-series: (i) the proportion of fully/overexploited stocks >1, (ii) industrialized or destructive fishing practices were in place, and (iii) there were documented community or ecosystem impacts caused by fishing. See text for further detail.

Ecosystem	Initial year	Criterion 1	Explanation	Source	Criterion 2	Explanation	Source	Criterion 3	Explanation	Source	Initial state
Baltic Sea	1974	0.67	–	H. Ojaveer (pers. comm.); IndiSeas WG; ICES (2008a, b)	Y	First attempts to introduce otter trawling in the early 1900s, rapid development and increase in numbers in the 1910s and 1920s. Breakthrough in the 1930s, when catch increased and this increase was attributed to bottom trawling	Bagge *et al.* (1994); Bager *et al.* (2007); Eero *et al.* (2007)	Y	Several studies have been performed in relation to this topic, but most of them originated after the 1990s. As similar fishing practices (netting and trawling) were in operation also in 1974, we assume that broadly similar impacts might have occurred too in 1974. Bottom trawling impacted demersal communities (including damage to several thin-shelled bivalve species and starfish) in the western Baltic. Bottom trawling has caused resuspension of sediments and dislocation of mainly epibenthic organisms, increase in predatory and scavenging species. Bottom trawling has especially impacted endofauna (than epifauna) and caused greater community evenness and diversity. Bottom trawling has caused remobilization of nutrients leading to the increase in nutrient concentrations and oxygen consumption. The flatfish and cod net fishery has caused mortality of long-tailed ducks and velvet scoters in the Gulf of Gdansk, which amounts to 10–20% of the total wintering population of these species in the area	Krost (1990); Krost *et al.* (1990); Rumohr and Krost (1991, and references therein); Stempniewicz (1994)	Impacted
Barents Sea	1984	0.83		E. Johannesen and AB (pers. comm.); IndiSeas WG; Gjøsæter (1995); Matishov *et al.* (2004)	Y	Trawling for demersal species since 1931 with diesel side trawl, with large stern trawlers since 1950, and with medium tonnage freezer trawlers since the 1990s	Matishov *et al.* (2004)	Y	Herring stock collapsed in 1960s through overfishing and the environment. Since the early 1970s, the herring stock has been grossly overexploited, which could have led to an imbalance in the state of the predator–prey relationships	Hamre (1994)	Impacted

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Ecosystem	Initial year	Criterion 1	Explanation	Source	Criterion 2	Explanation	Source	Criterion 3	Explanation	Source	Initial state
Bay of Biscay	1992	0.48		M-JR (pers. comm.); IndiSeas WG; Anon. (1991); Forest (2001)	Y	Discarding was high in the *Nephrops* fishery on the Grande Vasière and in other bottom-trawl fisheries. Mesh sizes were too small, and small fish were caught	Anon. (1991); Dardignac (1988)	Y	Large chondrichthyan species were lost in the first half of 20th century	Quéro and Cendrero (1996)	Impacted
Bering Sea, Aleutian Islands	1978	>0	Evidence suggests that yellowfin sole, Pacific ocean perch, and sablefish were "fully exploited to overfished" in the 1960s and 1970s	KA (pers. comm.), IndiSeas WG; Bakkala (1993); NRC (1996)	Y	Heavy industrial fishing by foreign fisheries since the mid-1950s, with associated discarding of bycatch	KA (pers. comm.); IndiSeas WG; Bakkala (1993); NRC (1996)	Y	Stellar sea cow was hunted to extinction, and the right whale was hunted almost to extinction. Many species of marine mammals were harvested for commercial purposes, the abundance of some was severely reduced	Bakkala (1993); NRC (1996)	Impacted
Eastern Scotian Shelf	1970	>0	Haddock, cod, silver hake, and other groundfish were fully to overexploited; catch controls introduced by ICNAF in early 1970s	AB (pers. comm.); IndiSeas WG; Halliday and Pinhorn (1996)	Y	Industrial fishing by foreign fisheries since the 1930s, with associated discarding of bycatch	Breeze (2002); Halliday and Pinhorn (1996)	Y	Extirpation of the walrus (*Odobenus rosmarus rosmarus*), the Atlantic grey whale (*Eschrichtius robustus*), severe reduction in grey seals, cod, and other groundfish	Halliday and Pinhorn (1996); Waring *et al.* (2004); Rosenberg *et al.* (2005)	Impacted
Guinean EEZ	1985	0.2	The coastal fish community, settling in the extended shallow area of the Guinean shelf, was considered as little exploited until 1985 (Domain, 1999). Guénette and Diallo (2004) note, however, that even if artisanal fishing developed only since 1984, there was formerly a foreign industrial fishing (by USSR fleets) operating on this shelf from 1971 and having access to the deeper part of the distribution area of these coastal species	D. Jouffre (pers. comm.); IndiSeas WG; Domain (1999); Guénette and Diallo (2004)	Y	Distant-water fleets operated in the EEZs of many West African ecosystems considered here, and Senegal and Mauritania are known to have been subjected to industrial exploitation for >50 years	Chavance *et al.* (2004)	N	No documented community or ecosystem impacts caused by fishing seem to be available in the literature concerning the period before 1985	D. Jouffre (pers. comm.); IndiSeas WG	Impacted
Irish Sea	1973	>0		J. L. Blanchard (pers. comm.); IndiSeas WG; Brander (1980)	Y	Trawling, dredging, discarding	Hill *et al.* (1999); Kaiser *et al.* (1996); Lindeboom and de Groot (1998); Ball *et al.* (2000)	Y	Long-term changes in benthic communities. Disappearance of common skate *Raia batis*	Brander (1980, 1981); Bradshaw *et al.* (2002)	Impacted

Mauritanian EEZ	1982	0.5		K. o. Mohamed Abdallahi (pers. comm.); IndiSeas WG; FAO (2006); Gascuel *et al.* (2009); IMROP (2007)	Y	Most fishing is by trawls. No dredging, no dynamite, no cyanide fishing, and no blast fishing	Gascuel *et al.* (2004); IMROP (2007)	~	Weak evidence of overexploitation of *Octopus vulgaris* and some sparids from 1982, and a decline in the overall abundance was reported shortly thereafter	FAO (2006); Gascuel *et al.* (2007)	Impacted
North-central Adriatic Sea	1975	>0		M. Coll (pers. comm.); IndiSeas WG; Arneri (1996); Jukic-Peladica *et al.* (2001); Vrgoc *et al.* (2004)	Y	Intense bottom trawling and beam trawling have developed in the area at least since 1975. Discarding is intense (mean of 30%) and a bycatch of seabirds, mammals, and reptiles is taken	Arneri (1996); Jukic-Peladica *et al.* (2001); Vrgoc *et al.* (2004)	Y	Ecosystem assessed as highly impacted by fishing in 1975 compared with 1990s and 2000s, by analysing the outputs of mass-balance models	Coll *et al.* (2007, 2009)	Impacted
Northeast United States	1963	0.64		J. S. Link (pers. comm.); IndiSeas WG; http://www.nefsc.noaa.gov/sos/; ICNAF (NAFO precursor) reports	Y	It has been estimated that Georges Bank is trawled ~2–3 times per year; mobile bottom tending gear can come into contact with sensitive hard substrata throughout	Auster and Langton (1999)	Y	There have been notable community shifts from gadid and pleuronectid groundfish to elasmobranchs to small pelagics	Link and Brodziak (2002); Link *et al.* (2006); Link (2007); Ecosystem Assessment Program (2009)	Impacted
North Sea	1983	0.91		J. L. Blanchard (pers. comm.); IndiSeas WG; ICES working group data from 2007	Y	Beam trawling, dredging, discarding, industrial fishing for sandeels	Kaiser and De Groot (2000)	Y	The Northeast Atlantic was being "fully fished" by 1983	Grainger and Garcia (1996)	Impacted
Northern Humboldt	1983	0.25	Although proportion of under-/moderately exploited stocks is close to 1, in the 1980s, the Peruvian sector of the Humboldt Current ecosystem was highly impacted by the fishery. The main species was depleted (anchoveta), whereas the second most important (sardine) was being scarcely exploited	E. Diaz (pers. comm.); IndiSeas WG; Pauly *et al.* (1987); Pauly and Palomares (1989)	Y	Discards of fish by hold capacity limitations have been recognized as an inherent part of the anchovy fishery. It was estimated in 9% of annual landings	Castillo and Mendo (1987)	Y	Since the 1960s, the collapse of bird populations and a large change in the relative abundance of the main species of guano birds were related to both combined environmental (successive *El Niño* events) and fishery effects	Tovar *et al.* (1987)	Impacted

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Ecosystem	Initial year	Criterion 1	Explanation	Source	Criterion 2	Explanation	Source	Criterion 3	Explanation	Source	Initial state
Portuguese EEZ	1981	0.5	–	M. F. Borges and LH (pers. comm.); IndiSeas WG; ICES (1981, 2007); http://www.ices.dk/datacentre/StdGraphDB.asp	Y	Portuguese industrialized trawl fleets since the 1970s (the initial state). In addition, Spanish trawlers operated off Portugal in the 1970s, and a Russian distant-water fleet operated offshore during the 1960s and 1970s	M. F. Borges (pers. comm.); IndiSeas WG	N	Corals were probably already damaged during the 1970/1980s, but there are no documents and only sporadic information on pieces of coral being taken by trawls	M. F. Borges (pers. comm.); IndiSeas WG	Impacted
Sahara coastal–Morocco	1993	>0	–	AB (pers. comm.); IndiSeas WG; Kifani *et al.* (2008)	Y	Distant-water fleets were already operating in Sahara coastal–Morocco waters by 1993	Kifani *et al.* (2008)	Y	Decrease in the trophic level of the catch and removal of large predators	Kifani *et al.* (2008)	Impacted
Senegalese EEZ	1981	0.6	–	D. Jouffre (pers. comm.); IndiSeas WG; Fontana and Weber (1982)	Y	Distant-water fleets operated in the EEZs of many West African ecosystems considered here, and Senegal and Mauritania have been subjected to industrial exploitation for >50 years. However, it is difficult to find documentation concerning really destructive fishing practice before 1981	Domain (1980); Chavance *et al.* (2004)	N	No documented community or ecosystem impacts caused by fishing seem to be available in the literature for the period before 1981	–	Impacted
Southern Benguela	1980	0.43	Several species were already considered to have been fully exploited by 1980. We estimate about half (57% of FAO species) were under- or moderately exploited in 1980 (cf. 28–35% moderately/underexploited in 2007)	LJS (pers. comm.); IndiSeas WG; local experts' knowledge (L. Hutchings, pers. comm.)	Y	Bottom trawls, discards (poorly quantified), dredging for oil, diamond mining. Trawling for *Pterogymnus laniarius* damaged deep-sea corals and other benthos	Cochrane *et al.* (2004)	Y	Documented changes in size structure of linefish and demersally trawled fish. Seabirds—more opportunistic species (benefiting from offal discards); competition with fisheries for prey	Crawford and Dyer (1995); Best *et al.* (1997); Crawford (1999); Crawford and Jahncke (1999); Crawford and Dyer (1995); Griffiths (2000); Yemane *et al.* (2004, 2010)	Impacted

Southern Catalan Sea	1978	>0	This is not certain because evaluations of the stocks were not fully available. However, a trawling ban was declared in the area to recover overexploited species from 1961 to 1966 (Plan Castellò). Anchovy was also heavily exploited	M. Coll (pers. comm.); IndiSeas WG; Lostado *et al.* (1999); Farrugio *et al.* (1993); Sardà (1998); Papaconstantinou and Farrugio (2000); Palomera *et al.* (2007)	Y	Intense bottom trawling and purse-seining have been developing in the area since the 1940s and 1950s. Discarding is intense (mean of 30%), and bycatches of seabirds, mammals, and reptiles are taken	Bas *et al.* (1955); Farrugio *et al.* (1993); Papaconstantinou and Farrugio (2000)	Y	The Plan Castellò of 1961–1966 was a ban on trawling owing to a reduction in the yield of commercial species in the whole area as a consequence of overfishing	Lostado *et al.* (1999); Coll *et al.* 2008b	Impacted
Southern Humboldt	1993	>0.5	–	SN (pers. comm.); IndiSeas WG	Y	Discarding in the trawl fisheries for Chilean hake (industrial fleet) and squat lobsters (small-scale fleet)	SN (pers. comm.); IndiSeas WG	Y	A significant decline in the mean trophic level of the landings from the late 1970s to the late 1990s has been reported. Top predators such as sea lions are at low levels compared with the years before the onset of industrialized fishing	Arancibia and Neira (2005)	Impacted
West coast Canada	1980	0.43	–	I. Perry (pers. comm.); IndiSeas WG	Y	Hard-bottom trawling. Particularly large biomasses of demersal species were removed by foreign fleets during the late 1960s and the 1970s. Time-series of fish catches from the 1920s for this region show catches of bottom species	Waddell and Ware (1995)	Y	An impact of trawling on (non-target) halibut stocks, dating from the early 1970s, and an impacts of shrimp trawling on bycatch species, especially eulachon. Reports of impacts documented during the 1990s, but similar effects were prevalent during shrimp trawling in the 1970s	Hoag (1971); Olsen *et al.* (2000)	Impacted

Appendix B

Comparison of trends in ecosystem indicators and attributes, and decision-tree diagnosis for three time-windows, pre-1980s to 2005, 1980–2005, and 1996–2005, under Decision Rule 1.

Ecosystem	EF (fish size)	SR (lifespan)	RP (biomass)	CB (% predators)	EF (trophic level)	RP (inverse fishing pressure)	Diagnosis*
Pre-1980s to 2005							
Baltic Sea	−		−				D
Bering Sea, Aleutian Islands	+						Not I
Eastern Scotian Shelf	−	−			−	+	D
Irish Sea	−		+	+	−	+	D
North-central Adriatic Sea	−	+	−			−	D
Northeast United States	−	+	+	−		−	D
Southern Catalan Sea	−	−				+	D
1980–2005							
Baltic Sea	−	−		−	−		D
Barents Sea						+	Not I
Bering Sea, Aleutian Islands	+	+					I
Eastern Scotian Shelf	−		−		−	+	D
Guinean EEZ	+			+	+	−	D
Irish Sea	+		+		−	+	D
Mauritanian EEZ					−	−	D
North-central Adriatic Sea	−	+	−			−	D
Northeast United States		+	+				I
North Sea	−		+		−	+	D
Northern Humboldt		−		+	+	−	D
Portuguese EEZ			−				D
Senegalese EEZ			−			−	D
Southern Benguela			+		−	+	D
Southern Catalan Sea	−						D
West coast Canada			+		+		I
1996–2005							
Baltic Sea			−				D
Barents Sea							Not I
Bay of Biscay							Not I
Bering Sea, Aleutian Islands							Not I
Eastern Scotian Shelf							Not I
Guinean EEZ		+					Not I
Irish Sea		+			−		D
Mauritanian EEZ							Not I
North-central Adriatic Sea						−	D
Northeast United States							Not I
North Sea						+	Not I
Northern Humboldt		−	+				D
Portuguese EEZ	+						Not I
Sahara coastal–Morocco			+		−	+	D
Senegalese EEZ			−				D
Southern Benguela		−	+	−	−		D
Southern Catalan Sea				+	+		I
Southern Humboldt	+				−		D
West coast Canada			+				Not I

+, significant positive trend; −, significant negative trend; blank, no trend; grey cells, no data. Note that it was assumed that missing indicators did not contribute to the diagnosis.

EF, ecosystem structure and functioning; SR, ecosystem stability and resistance to perturbations; RP, resource potential; CB, conservation of functional biodiversity.

*I, improving; Not I, not improving; D, deteriorating.

doi:10.1093/icesjms/fsp283