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

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Abstract:

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

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.,* 2008). Indeed a recent study has shown that there is now barely any part of the world's oceans that has not been impacted at some level through anthropogenic activity, be it fishing, pollution, shipping or eutrophication (Halpern *et al.,* 2008). In addition, we are living through a period of environmental change, the effects of which we are only beginning to explore (Hays *et al.*, 2005; Bender, 2007; ICES, 2008; Cheung *et al.*, 2009) and which are difficult to predict.

From a fisheries perspective, the old world of single species stock assessment and management is being replaced by a more holistic "ecosystem approach to fisheries" (or variations on the theme, FAO, 2003; Garcia *et al.,* 2003; Daan *et al.,* 2005; Pitcher *et al.,* 2008). EAF still includes single species stock assessment but is expanded to minimally include the wider impacts of fishing on the ecosystem, the role of the environment on species 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 enable an ecosystem approach to fisheries, a fundamental 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 (measures of management performance) and ii) control rules (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 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 stakeholders' interests (Cury *et al.,* 2005b). However, methods are being developed to overcome these difficulties, such as using historical or theoretical patterns to define references 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 the management decision-making process. This means that there may be no need to identify absolute reference points for the same indicators (Jennings and Dulvy, 2005): the more essential action is to stop the trend and reverse it.

There is no single indicator that can provide management with the information required for an EAF. Rather a suite of indicators that captures a range of impacts on the ecosystem, and its response is required, with the results synthesised 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; DFO 2003; Coll *et al.,* this volume; Link *et al.,* this volume) or a decision tree approach (Rochet *et al.,* 2005; Trenkel *et al.,* 2007).

Multi-criteria decision analysis has a wide and varied application in fisheries and resource management (Keeney and Raiffa, 1976; 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 visualise the components of a problem and to compare the choices that can be made. The basis of a decision theoretic approach is a decision tree, essentially a conceptual image (FAO, 2003) that portrays the problem as a tree, with the overall problem or objective at the top, and its various sub-components on the branches of the tree. In a classic application of the decision tree, each branch would terminate in a measurable objective (Keeney and Raiffa, 1976; FAO, 2003). Rochet *et al.* (2005) developed a "Decision Tree" approach to classify marine ecosystems into "improving", "stationary" or "deteriorating" based on the trends of key population and community indicators. That approach is adapted here to diagnose a broad range of ecosystems with respect to their initial condition (un-impacted or impacted).

The suite of ecosystem indicators estimated for the nineteen ecosystems included in this analysis were selected and calculated by the IndiSeas working group, 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 that uses a comparative approach to evaluate the effects of fishing on marine ecosystem (see other papers in this volume). A comparative approach is particularly useful in an ecosystem context since they can be treated as pseudo-replicates. It thus provides more confidence about the observed patterns of response to fishing seen across multiple ecosystems, rather than observations in one ecosystem alone.

Methods

The ingredients for the Decision Tree in this analysis is 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 building of a decision tree with decision rules.

The Ecosystems

The nineteen ecosystems examined in this study cover a broad geographical range, including the eastern Pacific (north, central and south); the north west Atlantic; the eastern Atlantic (north, central and south) and the Mediterranean Sea (Table 1). They include high latitude, temperate, tropical and upwelling systems and are associated with both developing and developed nations, and have varied fishing histories. Thus, 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.* (this volume).

Definition of Initial State

In order to interpret the results of the decision tree, it was necessary to first describe the state of the ecosystem at the beginning of the time period under consideration. Initial ecosystem state was simplified to a binary state: un-impacted or impacted. This was determined by asking the experts representing each of the 19 ecosystems to complete a survey where 3 criteria were used to assess the state of the ecosystem. An ecosystem was defined as un-impacted if all of the following criteria applied at the beginning of the time-series: (i) the proportion of under/moderately exploited stocks = 1 (i.e.., no overexploited stocks at the beginning of the time series), (ii) there were no industrialized or destructive fishing practices (e.g. trawling over hard bottoms; dredging; dynamite; discarding; cyanide fishing; blast fishing;) and (iii) there were no documented community or ecosystem impacts caused by fishing, such as: habitat loss, impact on by-catch species, disruption of the food web, loss of top predators. Respondents were asked to provide references to support their assessment (Appendix 1).

The Suite of Ecosystem Indicators

Rochet *et al.* (2005) used a combination of 2 population and 6 community indicators for their diagnostic analysis, although they only explicitly used community indicators for their decision tree. In contrast, only ecosystem indicators are used here. While a combination of population and community indicators may add robustness to the approach, the exclusion of population indicators enables a much broader comparison of ecosystems: a comparative approach requires that data for the indicators are available and operationally equivalent for all systems compared. This constrains the choice of indicators, and in some cases, the length of the time series. The suite of six indicators used here and the rationale associated with their choice are fully described in Shin *et al.* (this volume). Note that the IndiSeas WG uses the term “ecosystem” indicator while others such as Rochet *et al.* (2005) used the term “community” indicator. We use the wider term since the objective of IndiSeas is to apply a *suite* of indicators to assess the status of the ecosystem. Although the suite of ecosystem indicators presented here measures mostly the community sampled by demersal trawl surveys, and thus mostly fish, the longer term objective of IndiSeas is to expand this encompass the ecosystem, within the constraints noted above.

Each indicator is associated with a management goal (Table 2) and previous analyses show that the 6 indicators provide complementary information on the status of marine ecosystems (Blanchard *et al.,* this volume). 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 US), with a mean of 24 years (Table 1). One indicator was missing from the suite of four ecosystems: Sahara Coastal Morocco, West Coast Canada and northern Humboldt lacked fish size and Bay of Biscay lacked mean life span.

Determination of indicator trends

The information used in the decision tree is the trend of the indicators over the time period under consideration. For each indicator, we assume that a negative trend indicates increasing impacts of fishing (Table 2), 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, thus the indicator did not change, or 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 “un-impacted” ecosystem cannot be classified as improving since, by definition, there is no impacted state to improve on (in relation to fisheries impacts). Thus the basis of the classification is asymmetrical.

Trends in the indicators were examined following the method described in Coll *et al.* (2008) and used in Blanchard *et al.* (this volume) 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 (H0: = 0) using a two-tailed test of significance.

The Decision Tree

The indicators in the decision tree are assessed sequentially and decision rules developed to integrate this 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 rule. Thus in the decision tree, 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 Rochet *et al.,* 2005, Figure 2). This is a precautionary approach which insists that one negative trend in an ecosystem indicator indicates that the ecosystem is deteriorating, and this should invoke some sort of remedial management response. The requirement that two indicators must increase in order to be classified as improving is again conservative. Both are somewhat *ad hoc* and would require agreement between decision makers and stakeholders. This decision rule was adopted as Decision Rule 1.

The "one strike and you are out" rule of Rochet *et al.* (2005) was adopted here, but other decision rules were also explored in this 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 probability associated with each possible end point of the decision tree, under a null hypothesis of a stable community structure, was estimated as the product of the probabilities associated with each branch of the decision tree (assuming that the six ecosystem indicators are independent, see below). In a stable community, the probability of the null hypothesis being correct is 0.95 (assuming an error risk of 0.05 and using a two-tailed test of significance). Thus the probability of an increase, or decrease, is 0.025, and the probability of no change in indicator “a” followed by a decrease in indicator “b” is 0.024 (0.95*0.025). Missing indicators were assumed not to change and were thus attributed a probability of 0.95. The number of branches included in the probability estimation is determined by the point at which the tree is terminated by the decision rule: after the first significant negative trend, after the second significant positive trend providing there are no significant negative trends, or, at the end of the tree if no decision rule is invoked.

Indicator Order

The initial order of the indicators was determined following Rochet *et al.* (2005) who indicated 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 management goals outlined in Table 2 had to be represented in the first four indicators. To this end the order depicted in Table 2 was used. To check for sensitivity to starting order, five other variants that also satisfied all the criteria were explored. In each variant, each indicator was sequentially placed at the top of the decision tree, and the indicator that was at the top, placed at the bottom.

Interpreting trends

A key component of decision tree methodology is the interpretation of the trends of the ecosystem indicators. In all cases, the expected effect of fishing is a decrease in the indicator representing a negative condition. However, fishing does not occur in isolation, and other factors such as change in environmental conditions, increased productivity or increased recruitment may also affect a change in indicator trend. A list of factors that affect the direction of the six ecosystem indicators are proposed in Table 3. Furthermore, certain combinations of indicators may be indicative of specific effects of fishing or environmental forcing. Four effects were explored here: (i) fishing down the food web (FDFW), (ii) loss of diversity and stability (iii) loss of size structure and (iv) changes in productivity or recruitment (Table 4).

FDFW occurs when the average trophic level of the catch declines over time with a concurrent decrease in the total catch (Pauly *et al.,* 1998, 2001). Considering only the trophic level of catch can in some instances be misleading because it can reflect the high variability of the availability of small pelagic fish. Total catch is not included as an indicator here (see Shin *et al.,* this volume), but it is suggested that a combination of a significant negative trend in trophic level of the landings plus a significant negative trend in biomass would indicate FDFW and represents a departure from two of the management goals “Ecosystem structure and Functioning” and “Resource Potential” (Table 2). A decrease or no change in the other ecosystem indicators would also be consistent with FDFW.

Loss of diversity and stability is indicated by a decrease in % predators, indicating a loss in functional diversity (Hooper *et al.,* 2005) and mean life span, indicating a loss in stability. Stability is reduced since longer lived fish are more resistant to perturbation and change as, being larger, they are generally more fecund and have more viable eggs (Longhurst, 2002).

Loss of diversity and stability represents a departure from the management goals "Conservation of Biodiversity" and "Ecosystem Stability and Resistance to Perturbations" (Table 2). A decrease in predators represents a loss in functional diversity. A decrease or no change in trophic level of the landings, fish size or inverse exploitation would be consistent with this effect; no specific trend is expected for biomass.

Loss of size structure is the first potential indicator of size selective effects of fishing where large fish were targeted (Stokes *et al.*, 1993; Sinclair *et al.*, 2002). If coupled with a negative trend in mean life span and/or trophic level of the landings and/or % predators, this would be strong evidence of size selective fishing indicating a departure from the management goal "Ecosystem structure and Functioning". If fishing selectively targeted small fish, a significant positive trend would be expected in fish sizes. There is no expected trend in biomass, or inverse exploitation.

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 either increased productivity or recruitment. It is suggested that an increase in productivity or recruitment would be propagated up the food web through bottom-up processes, leading to increased productivity at all trophic levels or size groups. Thus it would have a transient impact on mean life span, % predators, trophic level of the landings and inverse exploitation. Therefore no change in these indicators is expected.

In addition to the classification function of the decision tree, the combinations of trends in the 19 ecosystems were examined to determine whether these processes could be detected by means of particular trend combinations and to determine whether there were other known factors that may contribute to the results.

Exploring changes in ecosystem status over time

The time window through which data are explored can affect the results that are seen. In order to explore whether the results observed for these ecosystems are consistent over time, the date were further explored in three time blocks: (i) 1960s/70s to end of the time series for those ecosystems with data (seven ecosystems, Table 1), (ii) 1980-end of time series (sixteen ecosystems, Table 1, also see Coll *et al.,* this volume) and (iii) 1996-2005 (nineteen ecosystems, Table 1, also see Blanchard *et al.,* this volume). Initial state in time periods (ii) and (iii) was determined by the results of the decision tree for time periods (i) and (ii) respectively for ecosystems whose original initial state was defined in earlier years (Table 1). Only Order 1 and Decision Rule 1 were used in the decision tree.

Independence of indicators

One assumption of the decision tree method is that the six indicators are independent. Blanchard *et al.* (this volume) conducted two sets of tests to explore redundancy in the indicators using a pair-wise correlation analysis and mutual information analysis which compares the rhythms of two time series to quantify their degree of dynamic cohesion

(Cazelles, 2004). Results indicated that there was no consistent redundancy between indicators across all 19 ecosystems and classification of ecosystems into groups according to pair-wise correlations of indicators resulted in fairly weak associations across ecosystems. Given these results, the assumption of independence between indicators holds for the time periods under consideration when all 19 ecosystems are considered.

Results

Initial State

The initial state of one ecosystem began in the 1960s, six in the 1970s, nine in the 1980s and three in the 1990s. No ecosystem met criterion 1, that is, the proportion of under or moderately exploited stocks = 1, and for at least 50% of the ecosystems, the proportion was less than 0.5. No ecosystem had no industrialized or destructive fishing practices but four had no documented community or ecosystem impacts caused by fishing (Guinea, Mauritania, Portugal and Senegal, Table 5). Thus all nineteen ecosystems were classified as impacted since they failed to meet at least one of the three criteria outlined above.

Determination of indicator trends

The analysis of trends in the ecosystem indicators over the entire length of their time series illustrates that most of these systems are undergoing change (Table 6). The Bay of Biscay had no ecosystem indicators with a significant trend for the time periods under consideration. West Coast Canada had two positive trends (biomass and trophic level of the landings), the Barents Sea and Bering Sea each had one positive trend (inverse fishing pressure and fish size respectively). In five cases, Baltic Sea, Mauritania, Portugal, Senegalese EEZ and Southern Humboldt, all significant trends were negative at the 5% level (note that there were positive trends in life span and fish size at the 10% level for Mauritania and the Baltic respectively). A mixture of positive and negative significant trends in the ecosystem indicators occurred in 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 and Figure 1b) and the rest had deteriorated from their initial state (Figure 1c and d). 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 Barents Sea and Bering Sea each had one positive trend (inverse exploitation and mean size respectively). The ecosystems that deteriorated from their initial state fell into four main groups: eight ecosystems were immediately classified as deteriorating because the first indicator at the top of the decision tree, mean length significantly decreased (Figure 1c); three ecosystems (Portugal, Senegal and Southern Humboldt), passed the first two levels of the decision tree, but had significant decreases in their biomass (Figure 1c); Mauritania, Southern Benguela and Sahara Coastal Morocco had significant decreases in mean trophic level of landings (although biomass increased in Southern Benguela and Sahara Coastal Morocco, Figure 1c) and in Guinea EEZ (Figure 1d), three indicators increased, but the last, inverse fishing pressure, decreased, eventually placing Guinea in the deteriorating class.

The probabilities associated with the end points of the decision tree in Figure 1 indicate that, under the null hypothesis of a stable community, the probability of observing these combined trends in indicators is extremely low.

Decision Rule 2 yielded very similar results, with the main difference that no ecosystem was classified as improving since there were no ecosystems with 3 significant increases and no negative trends (Table 6) and four ecosystems were re-classified as "not improving". Thus West Coast Canada, which had been improving under Decision Rule 1 was classified as "not improving" since it only had two positive trends. In addition Guinea EEZ, Southern Benguela, Portugal and Sahara Coastal Morocco were re-classified as not improving since they had only one negative trend. However, 11 ecosystems remained in the deteriorating category.

Effect of Indicator Order

Due to the precedence of negative trends in the decision rule (i.e. the decision tree stops when 1 or 2 negative trends are encountered) and the conditions under which an ecosystem can be classified as improving (i.e. 2 or 3 positive trends with no negative trends), the order of the indicators does not affect the final classification. For example, in the case of Guinea

EEZ, Figure 2, the length and number of branches in the tree varies with the order of the indicators, but the ecosystem is always classified as deteriorating. This is the case for all the ecosystems: the order of the indicators does not affect the classification of the ecosystem.

However, the order of the indicators does affect the probability associated with that outcome. In the case of Guinea EEZ for example (Figure 2), the probability of observing these trends under the null hypothesis of a stable community ranges from $p \cong 0.000$ (Order 4) to $p = 0.025$ (Order 6). Nevertheless, this quantitative difference does not translate into a qualitative difference: under any order of indicators (1 to 6), the probability of the trends found in Guinea occurring is low under the stability hypothesis.

The probabilities associated with the end points of the decision tree were estimated for all ecosystems for each of the six alternative orders of indicators (Table 7). The range of the values was small in all cases except for those ecosystems with a mixture of positive and negative trends, such as Guinea EEZ, Irish Sea, North Sea, Northern Humboldt, Southern Benguela and the Southern Catalan Sea (Tables 6 and 7). Regardless, and as in the case of Guinea EEZ which had the most extreme range of values, these were quantitative differences and did not affect the final classification.

Interpreting trends

Fishing down the food web was evident in the Southern Humboldt which was the only ecosystem with significant negative trends in both the trophic level of the catch and biomass (Table 8). Senegal and Portugal also showed signs of FDFW, with significant negative trends in trophic level and biomass although, the negative trend in trophic level of the catch was only significant at the 10% level. Senegal also had a significant negative trend in inverse exploitation. Five other ecosystems had significant negative trends in the trophic level of the catch, but in four cases this was coupled with an increase in biomass (Irish Sea, North Sea, Sahara Coastal Morocco and Southern Benguela).

Loss of diversity and stability was not observed in any system. Four ecosystems showed evidence of loss of size structure. The combination of indicators on the eastern Scotian Shelf and south Catalan sea (significant negative trends in fish size, lifespan and trophic level of the catch (Eastern Scotian Shelf only)) and in the North Sea (significant negative trends in fish size and trophic level) suggest that large fish have been targeted and selectively removed from the ecosystem. One ecosystem, Guinea, exhibited the opposite effect, with an increase in fish size, trophic level of landings and % predators): indicating that either small fish were being targeted, or are less abundant for other reasons. This is not likely to be an indicator of system recovery since there is no increase in biomass and there is a significant decrease in inverse fishing pressure.

The Northeast US (North East US), Irish Sea and North Sea all showed evidence of increased productivity or recruitment; a significant increase in biomass coupled with a significant decrease in fish size. No change was expected in the other indicators, but the Irish and North Sea also exhibited a decrease in trophic level of the catch and an increase in inverse fishing pressure. In the North East US, while mean length decreased, mean life span increased and proportion of predators and inverse fishing pressure decreased. In West Coast Canada, there could be increased productivity/recruitment but the lack of an indicator for mean size precludes this conclusion.

Management Goals

The ecosystem indicators are each associated with one of four management goals (Table 2). A significant negative trend in an indicator indicates that the management goal is not being met. Conversely, a positive trend indicates an improvement towards that goal, although it does not indicate how close the ecosystem is to achieving the goal. The negative trends in eight of the fourteen "deteriorating" ecosystems indicated that they were failing to meet more than one management goal. However, in some cases, such as north east US, the two ecosystem indicators associated with resource potential had opposite trends. Biomass increased and inverse fishing pressure decreased. Almost fifty percent of the significant negative trends of the ecosystem indicators were associated with the "Ecosystem Functioning" goal and most of the significant positive trends, as well as many of the significant negative trends, were associated with "Resource Potential" management goal (Table 6). The trend in % predators was significant in four ecosystems: it increased in Guinea, Irish Sea and Northern Humboldt, suggesting increasing Conservation of Biodiversity whereas it decreased in the North East US, indicating a loss of Conservation of Biodiversity. Trends in mean life span are associated with Ecosystem Stability and Resistance to Perturbations, which decreased in the Eastern Scotian Shelf, Northern

Humboldt and Southern Catalan Sea, but increased in North Central Adriatic and North East US.

Exploring changes in ecosystem status over time

Of the seven ecosystems with time series beginning prior to the 1980s, three, the Irish Sea, Baltic Sea and North Central Adriatic were consistently diagnosed as “deteriorating” (Table 9) regardless of the time period under consideration. The eastern Scotian Shelf and the Southern Catalan Sea were both defined as “deteriorating” until the mid-1990s. In the latter time period, there was no trend in the ecosystem indicators for the eastern Scotian Shelf (Appendix 2) suggesting that this ecosystem has reached some stability. The southern Catalan Sea was diagnosed as “improving” in the latter time period with significant positive trends in % predators and trophic level of landings. The Bering Sea changed from a stationary diagnosis, to “improving” to “stationary” over the three time periods. Finally the north east US was diagnosed as ‘deteriorating” over the whole time series, “improving” since the 1980s and ‘stationary” since the mid-1990s. Mean life span and biomass had positive significant trends in the first two time periods, and no negative trends occurred during the second time period.

Of the nine ecosystems whose time series began during the 1980s, the Barents Sea remained “non improving” from 1980s-2005 and 1996-2005 (Table 9). Four ecosystems, Guinea, Mauritania, North Sea and Portugal were diagnosed as “deteriorating” from 1980s-2005, but as “not improving” in the latter time period since there were no significant trends in any indicator. Senegal, Southern Benguela and Northern Humboldt “deteriorated” during both time periods. In the case of Senegal, biomass and inverse fishing pressure decreased from 1980s-2005, and biomass from 1996-2005; in Southern Benguela, mean trophic level of landings decreased from 1980s-2005, then from 1996-2005, mean life span, % predators and mean trophic level of landings decreased; in the Northern Humboldt system mean life span and inverse fishing pressure significantly decreased from 1980s-2005, and mean life span decreased from 1996-2005 (Appendix 2).

The time series of three ecosystems began in the early 1990s and their diagnosis did not change from the 1980-2005 time period to the 1996-2005 time period. Two were diagnosed as deteriorating (Sahara Coastal Morocco and Southern Humboldt) and Bay of Biscay was classified as “not improving” (Table 9). However, for the southern Humboldt, some indicators did change: during 1980-2005, biomass and trophic level of landings decreased whereas during 1996-2005 trophic level of landings decreased and fish size increased (Appendix 2).

During the first time period, six of seven ecosystems were classified as “deteriorating”, and one as “not improving”; in the second time period, fourteen of the nineteen ecosystems were “deteriorating”, two were “not improving” and three were “improving”. In the last time period (1996-2005) eight of nineteen ecosystems were deteriorating, ten as “not improving” and one as “improving”.

Interpreting trends through time

The four ecosystem effects identified in Table 4 were not consistently observed over the three time periods (Table 10), but all were exhibited. Most striking is the greater number of effects evident in the 1980-2005 period, compared to the seven systems for which there were data prior to 1980 and compared to all 19 ecosystems in the latest time period, 1996-2005. In the latter time period, loss of diversity and stability was the only effect identified and only in one system, Southern Benguela, due to a significant decrease in life span and % predators, supplemented by a significant decrease on TL of landings. Loss of size structure was the most common effect observed. During 1980-2005, it was evident in the Baltic Sea, Bering Sea, North Sea and Guinea. The combination of indicators suggested that size selective fishing was directed at small fish in Guinea and in the Bering Sea and at large fish in the other ecosystems. The latter effect was also seen in Southern Catalan Sea (1976-2005). The loss of diversity and stability was only observed in two ecosystems, Baltic Sea and Southern Benguela (see above) in two different time periods. FDFW was also seen only in three ecosystems, Eastern Scotian Shelf, Southern Humboldt and Portugal during 1980-2005. In the latter case, the decrease in trophic level of landings was significant at the 10% level.

Two ecosystems were consistent across the first two time periods: the Irish Sea showed evidence of increased productivity over the whole time series and since 1980, while the Eastern Scotian Shelf exhibited signs of size selective fishing of large fish over these two time periods and FDFW during 1980-2005. During the 1980-2005 time period the Baltic sea exhibited both loss of diversity and stability and size selective effects of fishing, which were not apparent since 1996.

Discussion

One of the strengths of a comparative approach is that patterns replicated across ecosystems provide more confidence that these patterns are real, than if they were observed in one ecosystem alone. Several very ugly patterns were replicated across the nineteen ecosystems analysed here using a decision tree approach: (i) all nineteen ecosystems were considered impacted at the beginning of their time series, (ii) since the initial state fifteen further deteriorated under Decision Rule 1, and eleven under Decision Rule 2, (iii) most failed on two or more management goals, (iv) fishing down the food web, loss of size structure, or loss of stability and resistance were evident in eleven ecosystems. This is not entirely unexpected since many of these ecosystems have a long history of exploitation, particularly in the North Atlantic and Mediterranean, and fishing does have impacts on the ecosystem (see contributions in Hollingworth, 2000). Indeed, the probability that any of these ecosystems is a stable, neutral community is small (with the exception of the Bay of Biscay). However, the consistency of the diagnosis provided here is worrisome. Given that all ecosystems departed from a state that was already impacted, we have relabeled our diagnoses as ugly (deteriorating), bad (not-improving, since the ecosystem is remaining in its impacted state, and good(ish), improving, since the direction is good, but the ecosystems are still likely to be highly impacted.

Essentially, these results indicate that eleven to fifteen of these ecosystems are in a more impacted state now than they were at the beginning of their time series, that is, they are in an "ugly" state. Furthermore, eight of those ecosystems were still deteriorating when examined over the last 10 years (96-05), five were stationary, having deteriorated, and thus still in a "bad" state, and only one, the Southern Catalan Sea, after almost two decades of deterioration, was finally classified as improving, good(ish), but see below. Therefore none of these ecosystems can be considered in a good state during any time period, although the Bering Sea, North East US and West Coast Canada have less negative diagnoses. By any measure, the results of this analysis are very concerning.

Geographically, the ecosystems in the north Pacific may be in better shape than the ecosystems from the Atlantic and Mediterranean: both the Bering Sea and West Coast Canada were diagnosed as improving during 1980-2005, and stationary since then. This may be attributed to good management practices, productive regime shifts or both. Link *et al.,* (this volume) identified environmental drivers as important in both of these ecosystems. In the southern Pacific, both the Southern and Northern Humboldt were diagnosed as "deteriorating": these are both upwelling systems and subject to strong environmental drivers (e.g., Chavez *et al.,* 2003, 2008; Montecinos *et al.,* 2003; Alheit and Ñiquen, 2004; Shannon *et al.,* 2008), which should be taken into consideration when interpreting the suite of ecosystem indicators, see below. Ecosystems from the North Atlantic and Mediterranean have a long exploitation history and were generally diagnosed as "deteriorating", or "not improving", from a previously deteriorated state, excluding Bay of

Biscay and Barents Sea. Furthermore, fishing down the food web, loss of size structure, or loss of stability and resistance were mainly observed in north Atlantic ecosystems. The Bay of Biscay and Barents Sea were classified as “not improving” for all relevant time periods. The time series for the Bay of Biscay was short and the lack of significant trends may be a consequence of this. The Barents Sea data exhibits strong cyclical patterns and is not well suited to treatment using linear regressions. Results for this system should be interpreted with caution. The good(ish) diagnosis for the Southern Catalan Sea over the most recent time period was due to an increase in % predators and trophic level of landings. However, these increased as a result of a large decline in small pelagic fish in the 1990s, and thus the ecosystem was not in an improved state as the indicators alone may suggest.

Time Frames

The use of different time frames for the analysis showed that the results of the analysis were sensitive to the time frame used and, for some systems, revealed an evolutionary process. The North East US for example, was originally diagnosed as deteriorating (1963-2005) then it improved (1980-2005) and during 1996-2005 was “not improving”. Overall, the shortest time frame of 10 years, 1996-2005, produced the least number of significant trends (20) whereas the longer time periods produced more than twice this number (see also Blanchard *et al.,* this volume). This result is expected since it has been suggested that at least 10-15 years of data is required to detect trends Nicholson and Jennings (2004), and Trenkel *et al.* (2007) suggest 20 or more years.

However, it is also suggested here that there is a second, process-oriented explanation for the lack of trends in the 10 year time frame for some ecosystems. Many of these ecosystems were diagnosed as deteriorating over the longer time period, and by the mid-1990s had reached a new more impacted state, after which they remained relatively stable. This is the case for the Eastern Scotian Shelf (Bundy, 2005) and likely the case for the North East US (Link *et al.,* 2002), North Sea, Guinea, and Mauritania.

Of greater concern is the result that over 50% of the fourteen ecosystems diagnosed as deteriorating during 1980-2005, were still diagnosed as deteriorating during 1995-2005 (Baltic Sea, Irish Sea, North Central Adriatic Sea, Southern Benguela, Northern Humboldt, Senegal, Sahara Costal Morocco and Southern Humboldt). The first three have a long history of fishing, but notably have fewer negative trends in the recent time periods than during 1980-2005 (Appendix 2), suggesting a decline in the rate of deterioration. The latter two time series began in the early 1990s, so the length of time over which they are diagnosed as deteriorating is less than for the other ecosystems, although this should still raise flags of concern. Southern Benguela and Northern Humboldt are upwelling systems, in which environmental drivers have a large influence on the abundance and distribution of marine organisms. Thus the ecosystem indicators must be interpreted in this context. In particular, many upwelling systems are characterized by large stocks of small pelagic fish (see Shannon *et al.,* 2008, and this volume for further details) whose dynamics are impacted by the environment as well as fishing. As seen below, this is the case of Northern Humboldt and Southern Benguela.

In the Northern Humboldt, where strong climatic, bottom-up forcing affects fish productivity on a large range of scales (Chavez *et al.*, 2008; Gutiérrez *et al.,* 2008) the environment rather than fishing pressure appears to be the main driver of ecosystem change (Bertrand *et al.*, 2004, 2008a, 2008b, 2008c; Taylor *et al.,* 2008; Shannon *et al.,* 2008). Of the indicators explored here, mean life span and inverse fishing pressure significantly decreased from 1980-2005, and mean life span decreased during 1996-2005 (note that there was no mean length indicator for this system), but biomass increased. Mean life span likely decreased due to factors associated with a change from warmer period in the 1980s ('El Viejo' *sensu* Chavez *et al.*, 2003) to a cooler, more productive conditions in the 1990s ('La Vieja' sensu Chavez *et al.*, 2003). The short-lived anchoveta increased ('full anchovy era' *sensu* Gutiérrez *et al.*, 2007), larger pelagic predators such as hake declined due to overfishing and adverse climatic conditions (Ballón *et al.*, 2008; Guevarra-Carrasco and Lleonart, 2008), and the biomass of the jumbo squid, a short lived predator, dramatically increased (Argüelles *et al.*, 2008). For further details see Shannon *et al.* (this volume). Thus for the recent time period, 1996-2005, the classification of the Northern Humboldt as 'deteriorating"should be questioned recognizing the influence of environmental change on the indicators, and the increase in biomass. Except for some species, in particular the hake, the impacts of fishing were less over the recent decade than they were over the longer time period, as noted above for the Eastern Scotian Shelf, North East US, North Sea, Guinea, Mauritania and Senegal.

The Southern Benguela system appeared to deteriorate further over the most recent time period: the number of significant negative trends increased from one in 1980-2005 (trophic level) to 3 during 1996-2005 (mean life span, % predators and trophic level). Biomass increased significantly in both periods. Generally, a result such as this may be due to a recruitment process or increased productivity at lower trophic levels since biomass increased while the other indicators decreased. However, if this was the case, a decrease in mean size would be expected, but this did not occur, even at the 10% risk level. In fact, unusually high stock sizes of small pelagic fish were observed off South Africa in the early 2000s, accounting for decreased life span, % predators and trophic level of the landed catch. The reason mean size did not also decrease is that this particular indicator was derived from a detailed demersal survey data for Southern Benguela, which was not easily combined or comparable to pelagic survey data that would reflect small pelagic fish abundance. This emphasizes the necessity for considering data sources and ecosystem characteristics on an ecosystem by ecosystem basis when interpreting indicator trends.

These results show some consistency with the ranking of ecosystems according to short- and long-term trends by Coll *et al.,* (this volume). Three ecosystems, Adriatic Sea, Southern Catalan Sea and Baltic Sea ranked as highly impacted in the majority of cases, as they did in the results presented here. Coll *et al.* (this volume) classified five ecosystems as “becoming more impacted” in the recent decade, compared to six when the full period 1980-2005 was examined, whereas the decision tree analysis classified eight ecosystems in the recent decade as deteriorating and fourteen ecosystems during 1980-2006. Further consistencies are apparent in the details: Coll *et al.*(this volume) classified the Eastern Scotian Shelf, Baltic Sea, Southern Catalan Sea, Senegal and Southern Humboldt as “becoming more impacted” in the long time period 1980-2005, whereas they improved to "moderately impacted" , or in the case of the Eastern Scotian Shelf, "becoming less impacted" according to trends for 1996-2005 only. Their results also indicated that some ecosystems had deteriorated in the last 10 years: the Adriatic, Southern Benguela, Mauritania, North Sea, and North East US ecosystems were classified as "moderately impacted" in terms of trends from 1980-2005, whereas they had deteriorated in the last decade to become classified as "becoming more impacted" for the period 1996-2005. While the results of Coll *et al.* (this volume) are derived from a relative ranking process in contrast to the tripartite classification used here, they largely confirm our results.

Detecting ecosystem effects through the combinations of trends

The observed ecosystem effects supported the diagnosis of the decision tree: of the fifteen ecosystems that were diagnosed as "deteriorating" during the longer time periods considered (pre-1980s – 2005 and 1980 – 2005), six exhibited size selective effects of fishing, two experienced FDFW, two experienced loss of diversity and stability and four showed evidence of increased productivity. Only one effect (loss of diversity and stability in Southern Benguela) was noted during the 10 year time frame of 1996-2005, which is explained by the result that fewer significant trends were detected during this shorter time period. Southern Benguela was also the one ecosystem with more significant trends in this time frame. Increased productivity or recruitment was observed in the Irish Sea and North Sea. These ecosystems have been heavily exploited for over a century and are not showing any signs of improvement, other than reduced exploitation and an increase in biomass. In the case of the Irish Sea, it has continued to deteriorate.

Some ecosystems exhibited contradictory trends. Mean life span increased in the North East US and North Central Adriatic for example, while fish size decreased. Using data from 1995-2005, Blanchard *et al,* (this volume) found that life span and fish size were positively correlated in some ecosystems but negatively correlated in others. In the case of North East US, this may be due to the fact that fisheries have been (for some time) catching larger organisms and many of the organisms that are not now primarily targeted have longer life spans. In the North Central Adriatic Sea and the Southern Catalan Sea, some indicators responded contrary to the expected trends due to increasing fishing effort because fishing was mainly targeting and depleting the small pelagic fish. In the Northern Humboldt system, life span decreased, while % predators and trophic level of the catch increased (and inverse exploitation decreased). Negative trends in life span and positive trends in trophic level are explained by the shift to a cooler anchovy regime (short lived, not low TL species) since mid 1990s as reflected in both biomass and landings whilst positive trends in % predators is explained by the outburst of giant squid biomass specially after El Niño 1997-1998. In the Southern Humboldt mean fish size increased, but trophic level of landings decreased over the last decade. Arancibia and Neira (2005) found a significant decline in the mean trophic level of landings off Southern Humboldt for the period 1978-2002, but Neira (2008) consequently determined that this decline was first due to “fishing through the food wed” *sensu* Essington *et al.* (2006) and real FDFW has been occurring in this system only since mid 1990s.

The decision tree approach to classifying ecosystems into three states, improving, stationary (non-improving) or deteriorating is attractive for its simplicity and thus more likely to be acceptable to stakeholders. However, there are several methodological issues involved in this approach that are subjective including the assessment of the initial state, the selection of ecosystem indicators, the order of the indicators and the choice of decision rule.

Assessment of initial state

All nineteen ecosystems were assessed as impacted in their initial year and most failed to meet all three of the criteria and could thus be considered highly impacted. The method for assessing the initial state of the ecosystems was very conservative and demanded that three criteria should be met in order for an ecosystem to be described as non-impacted. The criteria were selected to include the effects of fishing on species and the ecosystem and were intended to maximize comparability. No ecosystem met all three criteria. Assessing the initial state of ecosystems 20 plus years ago can be challenging since there was less interest in the ecosystem effects of fishing at that time and less documentation. In some ecosystems there is also simply less information available. Information for the first criteria and second criteria was available for most ecosystems, but the third criteria leaves room for error. The criterion was "there were no documented community or ecosystem impacts caused by fishing". However, lack of documentation does not equate with lack of impact, and in several cases there was no documentation for the earlier time period. This did not affect the assessment of initial state of any one ecosystem, given the binary classification used here. However, if a less conservative method was used, such as "degree of impact", criteria 3 would be more important.

716 Selection of ecosystem indicators

717 Link *et al.* (2002), Caddy (2004) and others have recommended using a broad range of
718 indicators to assess the status of ecosystems. Link (2005), Methratta and Link (2006), and
719 Rochet and Rice (2005) have also noted the need to select indicators using a set of criteria
720 that are germane for the management issues at hand in a given ecosystem. However, when
721 making comparative evaluations of marine ecosystems globally, a more limited set of
722 indicators must be selected since the data must be available for all ecosystems. The
723 implication of this is a compromise between an ideal set of indicators and a realistic set of
724 indicators. While it is not proposed that the set of indicators selected here is complete, it is
725 proposed as a minimum set. The rationale for the choice of indicators is discussed further in
726 Shin *et al.* (this volume). Ideally, each management goal would have at least two indicators
727 associated with it to account for different sensitivities of indicators to change, thus new
728 indicators are required for the management goals of "Stability and Resistance to
729 Perturbations" and "Conservation of Biodiversity". Potential indicators of the former could
730 include a measure of the trophic structure of the ecosystem such as the mean trophic level
731 of the community, or a measure of whether a change in exploitation reflects energy loss up
732 the food, captured by indicators such as the FiB index (Pauly *et al.,* 2001), or ratio of
733 pelagic to demersal fish in the community, and for the latter, species richness or number of
734 vulnerable or endangered species. There is no shortage of such potential indicators
735 (Murawski, 2000; Daan *et al.* 2005; Link, 2005; Fulton *et al.,* 2005); the challenge is to
736 select those that are generally applicable and related to these management objectives. A
737 second class of indicators that monitor that state of the environment, and thus flag potential

738 effects of the environment on the suite of indicators are required, especially for ecosystems
739 with large environmental fluctuations such as upwelling systems.
740
741 Of the six indicators used in this study, fewer significant trends occurred for mean life span
742 and % predators, both of which were derived from fisheries survey information, suggesting
743 that these are less sensitive to change than the other indicators that were catch-derived.
744 Inverse fishing pressure, a catch and survey derived indicator, resulted in significant trends
745 in over 50% of the ecosystems. However, this indicator requires careful consideration and
746 interpretation. The estimation of both catch and biomass is fraught with difficulties, ranging
747 from bias and misreporting of catch data (Patterson, 1998; Watson and Pauly, 2001) to
748 inconsistent survey protocols between years or across different fisheries within an
749 ecosystem. It was assumed here that surveys were consistent through time and that catch
750 was accurately reported. Catch may also be affected by management measures, so while
751 inverse fishing pressure does give an index of the proportion of biomass exploited, it is not
752 necessarily a good measure of the amount of fishing pressure that can be sustained by a
753 resource, since the management measures themselves become implicit in the indicator.
754 Inverse fishing pressure can be ambiguous; an increase may represent a decrease in
755 resource potential or it could represent a precautionary management measure to avoid this.
756 For this reason, inverse fishing pressure was placed at the bottom of the decision tree.
757 Auxilliary information in the form of additional indicators, such as total landings, total
758 landings at a given trophic level, or information about management actions and policy
759 might help elucidate the meaning of inverse fishing pressure. For the 19 ecosystems

examined here, it only determined the outcome of the decision tree once under Decision Rule 1 (Guinea) and twice under Decisions Rule 2 (Mauritania and Senegal).

Thus when interpreting the effect of this indicator, context must be considered. If it were not for inverse fishing pressure, Guinea would have been classified as "improving" under Decision Rules 1 and 2 since three indicators (LG, BpBs and TLc) had positive significant trends for the long-term time series. Inverse fishing pressure decreased during the first 10 years of the data series, and then remained relatively flat. This was reflected in the diagnosis for 1996-2005 which classified Guinea as "not improving" rather then deteriorating. Changes in trophic level of landings and inverse fishing pressure suggest that since 1995, the fishery was targeting higher trophic level species at a higher fishing mortality. This reflects the fishing history of this ecosystem: it has undergone rapid escalation of fishing effort, focused on coastal demersal species, over the last two decades, accounting for the increase in trophic level of the catch and the decrease of inverse fishing pressure (Lobry *et al.,* 2003). Exploitation in the offshore area has been more intense over a longer period of time (Sidibé *et al.,* 1999, cited in Laurans *et al.,* 2004). Overall, the history of exploitation in Guinea is less intense that in Senegal or Mauritania (Domalain *et al.,* 2004; Lobry *et al.,* 2003).

Order of Indicators

Under the decision rules used here, the order of the indicators does not affect the final classification, only the probability under the null hypothesis that they end up there. Thus, the method appears to be relatively robust and consistent. This is not a trivial result. Order

1 was chosen because the indicators at the top of the tree have a clearer association with likely causes of change in the indicator, they are estimated with relatively good precision and the first four indicators represent the four management goals outlined in Table 2. The first four indicators are also a more direct assessment of the ecosystem using fisheries independent survey information, whereas the last two, trophic level of landings and inverse fishing pressure, are derivatives of the act of fishing (i.e. pressure indicators as opposed to state indicators, see Degnbol and Jarre, 2004) even though they can also reflect non-fishery-related impacts such as natural outbursts of species at low trophic levels in response to eutrophication, or environmentally-induced changes in availability of certain species to fisheries.

However, if the decision rules were changed, the order of the indicators could have a larger impact on the outcome. For example, if an “improving” state was re-defined as occurring when 2 positive trends were encountered, at which point the procedure was stopped, then Guinea would be classified as improving under Order 1. However if the order of the indicators were changed, the Irish Sea, North Sea, North East US, Northern Humboldt and Southern Benguela could all be classified as improving. 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 widely accepted and acted upon by fisheries managers.

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 adverse 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 since the criteria to classify an ecosystem as “improving” were more stringent. Relaxing these rules, or making them less risk adverse 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 fourteen ecosystems classified as “deteriorating” under Decision Rule 1, only North Central Adriatic Sea and North East US 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.

The strong take home message from this precautionary 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 management goals defined in Table 2 are not being achieved. 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 (Bundy *et al.* 2008; Rice, 2008). Furthermore, the potential effects of the environment, particularly in upwelling ecosystems (Shannon *et al.,* this volume), need to be considered when interpreting these ecosystem indicators. Link *et al.* (this volume) explored drivers of ecosystem change, and Coll *et al.* (this volume) used biotic factors to rank ecosystems and biotic, abiotic and socioeconomic factors to explain these rankings. The results of all these studies point to the same conclusion: the management goals for the fisheries in these ecosystems are not being met.

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Table 1. List of the nineteen ecosystems considered in the decision tree classification of ecosystems. MFA= FAO Major Fishing Area, Div= FAO Division (adapted from Shin *et al.*, this volume), including the years for which data are available to estimate the ecosystem indicators.

Coastal ecosystem	**Geographical area**	**Surrounding countries**	**FAO fishing zones**	First year of time series	Last year of time series
Bering Sea, Aleutian Islands	NE Pacific	Alaska, US	MFA: 67	1978	2005
West coast Canada	E Central Pacific	Canada		1980	2005
Southern Humboldt	SE Pacific	Chile	MFA: 87, Div: 2.2.	1993	2004
Northern Humboldt	SE Pacific	Peru	MFA: 87, Div: 1.1, 1.2	1983	2005
Eastern Scotian shelf	NW Atlantic	Canada	MFA: 21, Div: 4V, 4W	1970	2005
North East US	NW Atlantic	US		1963	2005
Bay of Biscay	NE Atlantic	France	MFA: 27, Div: VIIIa, b	1994	2005
North Sea	NE Atlantic	UK, Norway, Denmark, Germany, Netherlands, Belgium	MFA: 27, Div: IVa,b,c, IIIa	1983	2005
Barents sea	NE Atlantic	Norway	MFA: 27, Div: I, IIb	1984	2005
Irish Sea	NE Atlantic	Ireland, UK	MFA: 27, Div: VIIa	1973	2005
Portuguese EEZ	NE Atlantic	Portugal	MFA: 27, Div: IXa	1981	2005
Baltic Sea	NE Atlantic	Germany, Estonia, Sweden, Poland, Russia, Lithuania, Latvia, Finland	MFA: 27, Div: IIId 25 to 29	1974	2005
Southern Catalan Sea	NW 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	E Central Atlantic	Morocco	MFA: 34, Div: 1.3	1998	2005
Senegalese EEZ	E Central Atlantic	Senegal	MFA: 34, Div: 3.12	1986	2005
Guinean EEZ	E Central Atlantic	Guinea	MFA: 34, Div: 3.13	1985	2005
Mauritania	E Central Atlantic	Mauritania		1982	2005
Southern Benguela	SE Atlantic	South Africa	MFA: 47, Div: 1.6, 2.1	1986	2005

Table 2. Suite of six ecosystem indicators for diagnosing ecosystem status with corresponding management objectives (L: length (cm), i: individual, s: species, N: abundance, B: biomass, Y: catch (tons), D=decline over time, RP = Resource Potential, EF = Ecosystem structure and Functioning, CB=Conservation of Biodiversity, SR = Ecosystem Stability and Resistance to Perturbations, (adapted from Shin *et al.*, this volume).

Indicators	Headline label	Source	Calculation, notations, units	Expected Trend under fishing pressure	Conservation Goals
Mean length of fish in the community	*fish size*	Fisheries Independent Surveys	$\overline{L} = \frac{\sum_i L_i}{N}$ (cm)	D	EF
Mean life span of fish in the community	*life span*	Fisheries Independent Surveys	$\overline{LS} = \frac{\sum_S (age_{\max} B_S)}{\sum_S B_S}$ (y^{-1})	D	SR
Total biomass of species in the community	*biomass*	Fisheries Independent Surveys	*B* (tons)	D	RP
Proportion of predatory fish in the community	*% predators*	Fisheries Independent Surveys	prop predatory fish= B predatory fish/B surveyed	D	CB
TL landings	*trophic level*	Commercial landings and estimates of trophic level (empirical and fishbase)	$\overline{TL}_{land} = \frac{\sum_s TL_s Y_s}{Y}$	D	EF
1/(landings /biomass)	*inverse fishing pressure*	Commercial landings	B/Y retained species	D	RP

Table 3. Summary of reasons why the six ecosystem indicators may increase or decrease

Indicator	Reasons for an increase	Reasons for an decrease	References
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 due to favourable environmental conditions • Decrease in abundance of stocks comprised of small-sized fish • Increase in abundance of stocks comprised of large-sized fish • Regime shift toward condition favouring large fish species	• Loss of larger fish from dominant populations • Strong recruitment success in dominant populations • Decrease in mean growth rates due to unfavourable environmental conditions • Increase in abundance of stocks comprised of small-sized fish • Decrease in abundance of stocks of large-sized fish • Regime shift toward condition favouring small pelagic fish	Shin *et al.*, 2005; Rochet and Trenkel, 2005; Chavez *et al.*, 2003, 2008
Mean life span 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 life span • Regime shift toward condition favouring large fish species	• Decrease in the biomass of species with high longevity • Increase in the biomass of species with short life span • Demographic explosions of invertebrates (e.g. Cephalopods, crustaceans) • Regime shift toward 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 due to fishing • Good reproductive success leading to high recruitment due to a large spawner stock biomass, favourable climatic conditions, etc... • Reduced predation or increased food supply i.e. Favourable trophic interactions.	• Increased mortality – both fishing and natural (due to predation, pollution, unfavourable environmental conditions...). Fishing mortality can be of two kinds - direct and indirect. Direct mortality is the catch of target species, and includes the unintended mortality of non-target species, bycatch species and discarding. Indirect mortality includes increased predation or	Duplisea *et al.,* 1997; Rice, 2005; Rochet and Trenkel, 2005; Rogers and Greenaway, 2005

		competition propagated through the food web.as a result of species changes due to fishing. ● Poor reproductive success due to unfavourable environmental conditions, leading to low recruitment. ● Increased predation, increase competition, or depleted food supply i.e. Unfavourable trophic interactions	
% 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 toward 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 toward 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 • Targetting of larger individuals at higher tls within a species. • Regime shift toward condition favouring large fish species	• Fishing effort shifted to lower trophic levels due to: ○ 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	Pauly *et al.,* 1998a,1998b; Caddy *et al.* 1998; Essington *et al.*, 2006

		species and consequent decrease in trophic feeding level ○ Increase in availability or market value of commercially valuable invertebrates ○ Regime shift toward condition favouring small pelagic fish •
Inverse fishing pressure This indicator is a proportion, and change will be a result of change in landed catch, biomass or both. Furthermore, if both catch and biomass change in the same direction, exploitation rate may not change	May be due to an decrease in catch, a increase in biomass or both Decrease in landed catch due to: • 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 Biomass See Biomass row above in same table	May be due to an increase in catch, a decrease in biomass or both Increase in landed catch due to: • 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….) Decrease in Biomass See Biomass row above in same table

Table 4. Expected combination of trends of ecosystem indicators that specify a given effect on the ecosystem. RP = Resource Potential, EF = Ecosystem structure and Functioning, CB=Conservation of Biodiversity, SR = Ecosystem Stability and Resistance to Perturbations,

Effect on the ecosystem	fish size	life span	biomass	% predators	trophic level	Inverse fishing pressure	Conservation Goals Affected
Fishing Down the Food Web	↓ →	↓ →	↓	↓ →	↓	↓ →	EF, RP (SR, CB)
Loss of diversity and stability	↓ →	↓	↓ →	↓	↓ →	↓ →	SR, CB (EF, RP)
Loss of size structure[1]	↓ or ↑	↓ or ↑	—	↓ or ↑	↓ or ↑	—	EF (RP, SR)
Increased Productivity or recruitment	↓	—	↑	—	—	—	EF, RP

↓ = significant negative trend, ↑ = significant negative trend, → = no trend, — = no expected trend; a large arrow means that the indicator must be included in the trend combination; arrows in grey indicate other indicators and the direction of their slope that would be consistent with the effect on the ecosystem. For For more details, please see the text.

[1] All arrows should either point up or down.

Table 5. Results of the assessment of initial state of the nineteen ecosystems evaluated in this paper. An ecosystem was defined as un-impacted if all of the following criteria applied at the beginning of the time-series: (i) the proportion of under/moderately exploited stocks = 1 (ii) there were no industrialized or destructive fishing practices and (iii) there were no documented community or ecosystem impacts caused by fishing, see text for further details.

	Year for which initial state is described	proportion of under or moderately exploited stocks	no industrialized or destructive fishing practices?	no documented community/ ecosystem impacts caused by fishing?	Degree of impact (# Criteria not met)	Initial State
Baltic Sea	1974	0.33			3	Impacted
Barents Sea	1984	0.17			3	Impacted
Bay Biscay	1992	0.52			3	Impacted
Bering Sea, Aleutian Islands	1978	<1				Impacted
Eastern Scotian shelf	1970	<1				Impacted
Guinean EEZ	1985	0.8		√	2	Impacted
Irish Sea	1973	<0.33				Impacted
Mauritania	1982	0.5		√	3	Impacted
North Central Adriatic Sea	1975	<1			3	Impacted
North East US	1963	0.36			3	Impacted
North Sea	1983	0.09			3	Impacted
Northern Humboldt	1983	0.75			3	Impacted
Portuguese EEZ	1981	0.5		√	0	impacted
Sahara Coastal - Morrocco	1993	< 1			3	Impacted
Senegalese EEZ	1981	0.4		√	2	Impacted
Southern Benguela	1980	0.57			3	Impacted
Southern Catalan Sea	1978	<1			3	Impacted
Southern Humboldt	1993	< 0.5			3	Impacted
West coast Canada	1980	0.57			3	Impacted

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Table 6. Trends in ecosystem indicators over the entire length of their time series (to 2005) for the nineteen ecosystems evaluated in this paper using generalised least squares and autoregressive error. Significance levels are shown by bold (α=0.05) and italics (α=0.1) font.

	Ecosystem Indicator	Fish size	Life span	Biomass	% predators	Trophic level	Inv. fishing pressure	Length of time series (years)
	Conservation Goal	EF	SR	RP	CB	EF	RP	
	Ecosystem							
1	Baltic Sea	**-0.084**	*-0.057*	**-0.078**	-0.043	-0.047	-0.047	32
2	Barents Sea	0.072	0.053	*0.087*	-0.020	0.081	**0.070**	22
3	Bay of Biscay	0.063	-	-0.129	0.052	0.018	-0.027	12
4	Bering Sea, Aleutian Islands	**0.089**	0.001	0.022	0.023	0.038	-0.024	28[1]
5	Eastern Scotian shelf	**-0.071**	**-0.050**	-0.041	-0.021	**-0.045**	**0.078**	36
6	Guinea ZEE	**0.106**	0.039	-0.010	**0.101**	**0.093**	**-0.124**	21
7	Irish Sea	**-0.118**	*0.110*	**0.043**	**0.046**	**-0.091**	**0.079**	33[2]
8	Mauritania	*0.075*	0.044	0.045	0.010	**-0.144**	**-0.157**	24[3]
9	North Central Adriatic Sea	**-0.111**	**0.090**	**-0.069**	*0.061*	-0.033	**-0.070**	30[4]
10	North East US	**-0.044**	**0.058**	**0.056**	**-0.050**	0.012	**-0.051**	43
11	North Sea	**-0.130**	0.010	**0.082**	0.038	**-0.065**	**0.145**	23
12	Northern Humboldt	**-**	**-0.119**	*0.075*	**0.085**	**0.133**	**-0.079**	23
13	Portuguese ZEE	0.000	0.033	**-0.068**	0.032	*0.065*	0.040	25
14	Sahara Coastal - Morrocco	-	0.003	**0.366**	-0.230	**-0.310**	**0.321**	8
15	Senegalese ZEE	-0.054	0.039	**-0.132**	0.041	*-0.082*	**-0.141**	20
16	Southern Benguela	0.020	-0.078	**0.167**	-0.043	**-0.080**	**0.158**	20
17	Southern Catalan Sea	**-0.122**	**-0.089**	**0.051**	-0.046	0.025	**0.059**	26
18	Southern Humboldt	-0.055	0.066	**-0.193**	0.058	**-0.196**	0.063	12
19	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	

[1] Time series for fish size was 24 years long.
[2] Time series for fish size and lifespan was 16 years long.
[3] Time series for fish size was 20 years long and trophic level and Inverse fishing pressure was 16 years long.
[4] Time series for fish size, life span and % predators was 24 years long.

Table 7. The probabilities of ending up at the end points of the decision tree under the null hypothesis of a stable neutral community, under six alternative orders of ecosystem indicators and using Decision Rule 1 (see below for orders).

	Order 1	Order 2	Order 3	Order 4	Order 5	Order 6	Average	Range
Baltic Sea	0.025	0.024	0.025	0.021	0.023	0.024	0.024	0.004
Barents Sea	0.019	0.020	0.020	0.020	0.020	0.020	0.020	0.001
Bay of Biscay	0.735	0.735	0.735	0.735	0.735	0.735	0.735	0.000
Bering Sea	0.019	0.019	0.019	0.019	0.019	0.019	0.019	0.000
Eastern Scotian shelf	0.025	0.025	0.023	0.024	0.025	0.001	0.020	0.024
Guinea ZEE	0.000	0.000	0.000	0.000	0.001	0.025	0.004	0.025
Irish Sea	0.025	0.000	0.000	0.001	0.025	0.001	0.009	0.025
Mauritania	0.020	0.021	0.023	0.024	0.025	0.025	0.023	0.005
North Central Adriatic Sea	0.025	0.001	0.025	0.023	0.024	0.025	0.020	0.024
North East US	0.025	0.000	0.001	0.025	0.024	0.025	0.017	0.025
North Sea	0.025	0.001	0.001	0.024	0.025	0.001	0.013	0.024
Northern Humboldt	0.024	0.025	0.000	0.000	0.001	0.025	0.012	0.025
Portuguese ZEE	0.023	0.024	0.025	0.019	0.020	0.021	0.022	0
Sahara Coastal - Morrocco	0.001	0.001	0.001	0.024	0.025	0.001	0.008	0.024
Senegalese ZEE	0.023	0.024	0.025	0.023	0.024	0.025	0.024	0.002
Southern Benguela	0.001	0.001	0.001	0.024	0.025	0.000	0.008	0.025
Southern Catalan Sea	0.025	0.025	0.000	0.001	0.001	0.001	0.009	0.025
Southern Humboldt	0.023	0.024	0.025	0.024	0.025	0.021	0.024	0.004
West Coast Canada	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0

Order 1 = fish size, life span, biomass, % predators, trophic level, inverse fishing pressure
Order 2 = life span, biomass, % predators, trophic level, inverse fishing pressure, fish size
Order 3 = biomass, % predators, trophic level, inverse fishing pressure, fish size, life span
Order 4 = % predators, trophic level, inverse fishing pressure, fish size, life span, biomass
Order 5 = trophic level, inverse fishing pressure, fish size, life span, biomass, % predators
Order 6 = inverse fishing pressure, fish size, life span, biomass, % predators, trophic level

Table 8. Potential effects on the ecosystem during the entire time series (to 2005), under Decision Rule 1. FDFW = Fishing down the food web, DS = loss of diversity and stability, SS = size selective effects of fishing, PR = increased productivity or recruitment. Only ecosystems with observed effects are shown. Shaded cells indicate an effect; stippled cells indicate an effect where one or more of the significant trends were significant at the 10% level.

	FDFW	SR	SS	PR
Baltic Sea				
Barents Sea				
Bay of Biscay				
Bering Sea				
Eastern Scotian shelf				
Guinea ZEE				
Irish Sea				
Mauritania				
North Central Adriatic Sea				
North East US				
North Sea				
Northern Humboldt				
Portuguese ZEE				
Sahara Coastal - Morrocco				
Senegalese ZEE				
Southern Benguela				
Southern Catalan Sea				
Southern Humboldt				
West Coast Canada				

Table 9. Comparison of decision tree diagnoses for three time windows, pre-1980s-2005, 1980-2005 and 1996-2005, under Decision Rule 1. Shaded cells indicated diagnosis. I =improving, Not I = not improving, D = deteriorating.

		Pre-1980s-2005			1980-2005			1996-2005		
Start of time series	Ecosystem Diagnosis	I	Not I	D	I	Not I	D	I	Not I	D
pre-1980s	Baltic Sea									
	Bering Sea, Aleutian Islands									
	Eastern Scotian shelf									
	Irish Sea									
	North Central Adriatic Sea									
	North East US									
	Southern Catalan Sea									
1980s	Barents Sea									
	Guinea ZEE									
	Mauritania									
	North Sea									
	Northern Humboldt									1
	Portuguese ZEE									
	Senegalese ZEE									1
	Southern Benguela									
	West Coast Canada									
1990s	Bay of Biscay									
	Sahara Coastal - Morocco									
	Southern Humboldt									

Table 10. Comparison of potential effects on the ecosystem during three time windows, pre-1980s-2005, 1980-2005 and 1996-2005, under Decision Rule 1. FDFW = Fishing down the food web, DS = loss of diversity and stability, SS = loss of size structure, PR= increased productivity or recruitment. Only ecosystems with observed effects are shown. Shaded cells indicate an effect; stippled cells indicate an effect where one or more of the significant trends were significant at the 10% level.

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												
	North East US												
	Southern Catalan Sea												
1980s	Southern Benguela												
	North Sea												
	Guinea ZEE												
	Portuguese ZEE												
1990s	Southern Humboldt												

Figure Captions

Figure 1. Decision Tree Diagnoses for the nineteen ecosystems (a) Improving, (b) Not-Improving and (c) and (d) Deteriorating. The first node of the decision tree is Fish size. If this is decreasing, Decision Rule 1 is invoked and the decision tree is terminated (1c): if it is increasing or there is no trend, then the decision tree moves down to the next node (indicator), and same process is followed until either Decision Rule 1 terminates the decision tree, or the last node, inverse fishing pressure is reached. The 3 arrows at each node indicate the 3 possible directions of the indicator trend, decreasing (red), increasing (green) or no trend (striped); the thick black arrow indicates the observed trend. Thickness of branches connecting nodes represent the probability under the null hypothesis of a stable neutral community associated with that path: thick solid line (no change), p=0.95, thin dashed line (significant trend), p=0.025. The overall probability of the end-point of the decision tree for each ecosystem is the product of the probabilities associated with each node until the decision tree is terminated. NB. Note in 1(a) white nodes indicates missing data, where it is assumed p=0.95. See text for further details.

Figure 2. Decision Tree results for Guinea EEZ using Decision Rule 1, but under different indicator orders (see Table 3). In each case, Guinea EEZ is diagnosed as deteriorating, but the end probabilities associated with each result differ. For “Order 1” see Figure 1(d). See text and Figure 1for more details.

Figure 1.

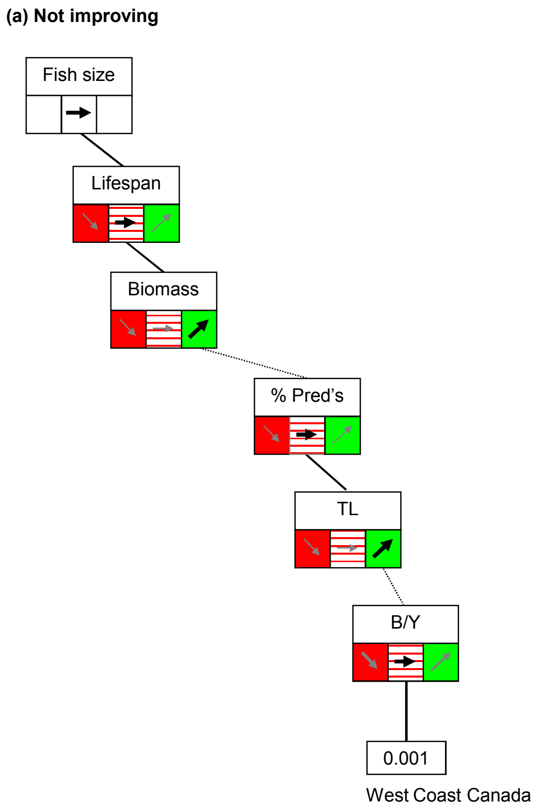

Figure 1a

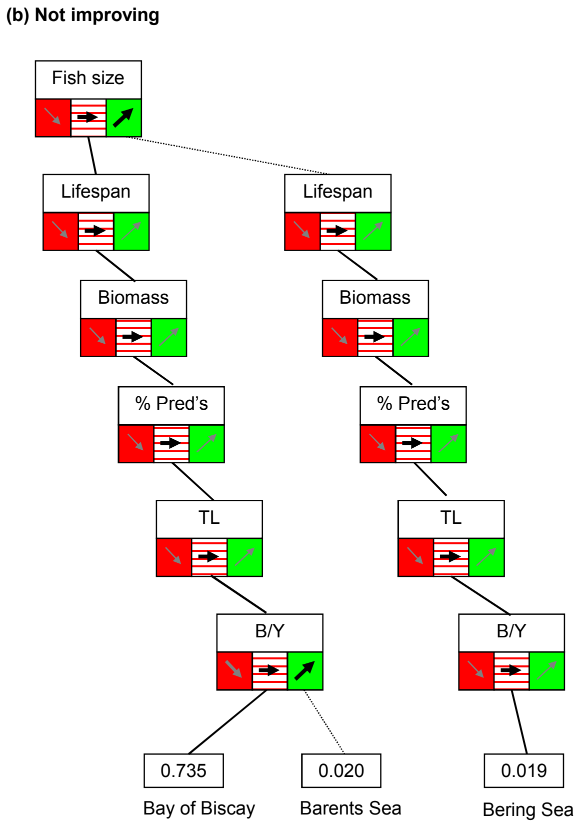
(b) Not improving
Fish size
Lifespan
Lifespan
Biomass
Biomass
% Pred's
% Pred's
TL
TL
B/Y
B/Y
0.735
Bay of Biscay
0.020
Barents Sea
0.019
Bering Sea

Figure 1b

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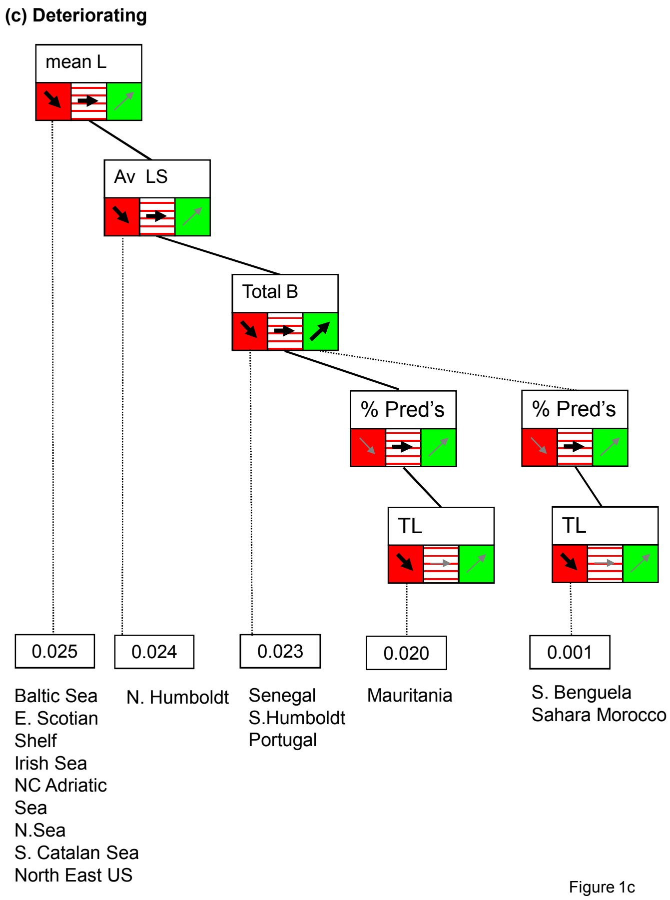

(c) Deteriorating
mean L
Av LS
Total B
% Pred's
% Pred's
TL
TL
0.025
0.024
0.023
0.020
0.001
Baltic Sea
E. Scotian
Shelf
Irish Sea
NC Adriatic
Sea
N.Sea
S. Catalan Sea
North East US
N. Humboldt
Senegal
S.Humboldt
Portugal
Mauritania
S. Benguela
Sahara Morocco

Figure 1c

45

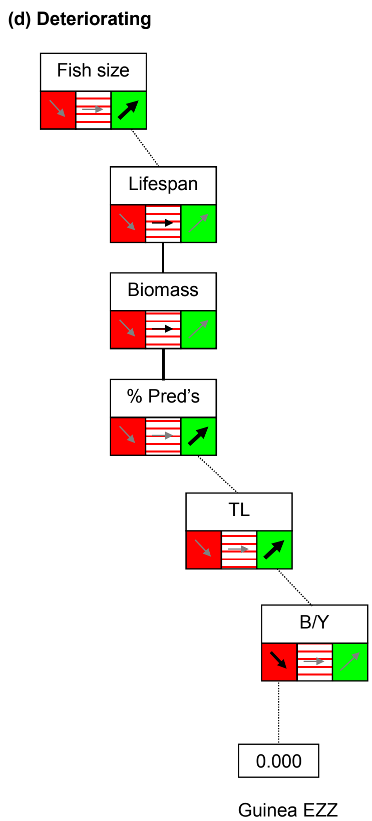
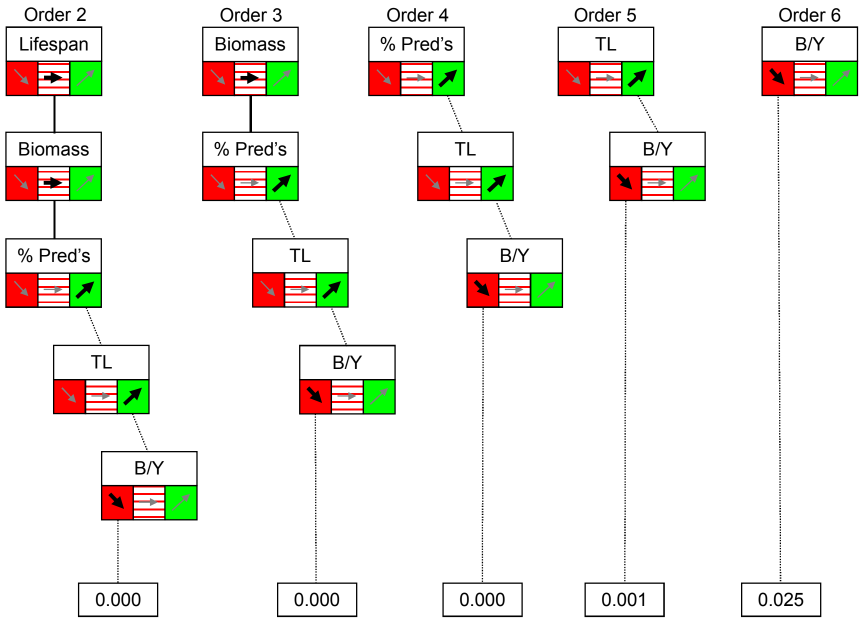

(d) Deteriorating
Fish size
Lifespan
Biomass
% Pred's
TL
B/Y
0.000
Guinea EZZ

Figure 1d

46

47 Figure 2.



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Appendix 1. Detailed Results for Initial State.
I have not been able to assemble this yet – to be added if editor and reviewers agree.

Appendix 2. Comparison of trends in ecosystem indicators for three time windows, pre-1980s-2005, 1980-2005 and 1996-2005, under Decision Rule 1, with decision tree diagnosis and the probabilities associated with each result. Green cell = significant positive trend, orange cell = significant negative trend, no shading = no trend, grey cells indicate no data. Note it was assumed that missing indicators did not to change and were assigned d a probability of 0.95. Overall probability is the product of the probabilities for each indicator until the decision tree is terminated by Decision Rule 1.

Ecosystem	EF mLG	SR mLS	RP Bs	CB BpBs	EF TLc	RP 1/LtBs	Diagnosis	Probability
Pre 1980s-2005								
Baltic Sea	0.025	0.95	0.025	0.95	0.95	0.95	D	0.025
Bering Sea, Aleutian Islands	0.025	0.95	0.95	0.95	0.95	0.95	Not I	0.019
Eastern Scotian shelf	0.025	0.025	0.95	0.95	0.025	0.025	D	0.025
Irish Sea	0.025	0.95	0.025	0.025	0.025	0.025	D	0.025
North Central Adriatic Sea	0.025	0.025	0.025	0.95	0.95	0.025	D	0.025
North East US	0.025	0.025	0.025	0.025	0.95	0.025	D	0.025
Southern Catalan Sea	0.025	0.025	0.95	0.95	0.95	0.025	D	0.025
1980-2005								
Baltic Sea	0.025	0.025	0.95	0.025	0.025	0.95	D	0.025
Bay Biscay	0.95	0.95	0.95	0.95	0.95	0.95	Not I	0.735
Barents Sea	0.95	0.95	0.95	0.95	0.95	0.025	Not I	0.019
Bering Sea, Aleutian Islands	0.025	0.025	0.95	0.95	0.95	0.95	I	0.001
Eastern Scotian shelf	0.025	0.95	0.025	0.95	0.025	0.025	D	0.025
Guinea ZEE	0.025	0.95	0.95	0.025	0.025	0.025	D	0.025
Irish Sea	0.025	0.95	0.025	0.95	0.025	0.025	D	0.025
Mauritania	0.95	0.95	0.95	0.95	0.025	0.025	D	0.950
North Central Adriatic Sea	0.025	0.025	0.025	0.95	0.95	0.025	D	0.025
North East US	0.95	0.025	0.025	0.95	0.95	0.95	I	0.001
North Sea	0.025	0.95	0.025	0.95	0.025	0.025	D	0.025
Northern Humboldt	0.95	0.025	0.95	0.025	0.025	0.025	D	0.024
Portuguese ZEE	0.95	0.95	0.025	0.95	0.95	0.95	D	0.735
Sahara Coastal - Morrocco	0.95	0.95	0.025	0.95	0.025	0.025	D	0.950
Senegalese ZEE	0.95	0.95	0.025	0.95	0.95	0.025	D	0.950
Southern Benguela	0.95	0.95	0.025	0.95	0.025	0.025	D	0.950
Southern Catalan Sea	0.025	0.95	0.95	0.95	0.95	0.95	D	0.025
Southern Humboldt	0.95	0.95	0.025	0.95	0.025	0.95	D	0.950
West Coast Canada	0.95	0.95	0.025	0.95	0.025	0.95	I	0.001
1996-2005								
Baltic Sea	0.95	0.95	0.025	0.95	0.95	0.95	D	0.023
Barents Sea	0.95	0.95	0.95	0.95	0.95	0.95	Not I	0.774
Bay Biscay	0.95	0.95	0.95	0.95	0.95	0.95	Not I	0.774
Bering Sea	0.95	0.95	0.95	0.95	0.95	0.95	Not I	0.774

Eastern Scotian Shelf	0.95	0.95	0.95	0.95	0.95	0.95	Not I	0.774
Guinea EEZ	0.95	**0.025**	0.95	0.95	0.95	0.95	Not I	0.019
Irish Sea	0.95	**0.025**	0.95	0.95	**0.025**	0.95	D	0.001
Mauritania	0.95	0.95	0.95	0.95	0.95	0.95	Not I	0.774
North Central Adriatic Sea	0.95	0.95	0.95	0.95	0.95	**0.025**	D	0.774
North East US	0.95	0.95	0.95	0.95	0.95		Not I	0.774
North Sea	0.95	0.95	0.95	0.95	0.95	**0.025**	Not I	0.774
Northern Humboldt	0.95	**0.025**	**0.25**	0.95	0.95	0.95	D	0.024
Portuguese ZEE	**0.025**	0.95	0.95	0.95	0.95	0.95	Not I	0.020
Sahara Coastal - Morrocco	0.95	0.95	**0.025**	0.95	**0.025**	**0.025**	D	0.001
Senegalese ZEE	0.95	0.95	**0.25**	0.95	0.95	0.95	D	0.774
Southern Benguela	0.95	**0.025**	**0.025**	**0.025**	**0.025**	0.95	D	0.950
Southern Catalan Sea	0.95	0.95	0.95	**0.025**	**0.025**	0.95	I	0.001
Southern Humboldt	**0.025**	0.95	0.95	0.95	**0.025**	0.95	D	0.001
West Coast Canada	0.95	0.95	**0.025**	0.95	0.95	0.95	Not I	0.020