Using indicators for evaluating, comparing, and communicating the ecological status of exploited marine ecosystems. 2. Setting the scene

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Shin, Y-J., Shannon, L. J., Bundy, A., Coll, M., Aydin, K., Bez, N., Blanchard, J. L., Borges, M. F., Diallo, I., Diaz, E., Heymans, J. J., Hill, L., Johannesen, E., Jouffre, D., Kifani, S., Labrosse, P., Link, J. S., Mackinson, S., Masski, H., Möllmann, C., Neira, S., Ojaveer, H., ould Mohammed Abdallahi, K., Perry, I., Thiao, D., Yemane, D., and Cury, P. M. 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.

Background is provided to the selection of ecological indicators by the IndiSeas Working Group, and the methodology adopted for analysis and comparison of indicators across exploited marine ecosystems is documented. The selected indicators are presented, how they are calculated is explained, and the philosophy behind the comparative approach is given. The combination of selected indicators is intended to reflect different dynamics, tracking processes that display differential responses to fishing, and is meant to provide a complementary means of assessing marine ecosystem trends and states. IndiSeas relied on inputs and insights provided by the local experts from participating ecosystems, helping to understand state and trend indicators and to disentangle the effect of other potential ecosystem drivers, such as climate variability. This project showed that the use of simple and available indicators under an ecosystem approach can achieve a real, wide-reaching evaluation of marine ecosystem status caused by fishing. This is important because the socio-economics of areas where fishing activities develop differs significantly around the globe, and in many countries, insufficient data are available for complex and exhaustive analyses.

Keywords: comparative approach, ecological indicators, ecosystem effects of fishing, exploited marine ecosystems, IndiSeas.

Received 19 June 2009; accepted 10 December 2009; advance access publication 8 February 2010.

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Introduction

Over the past two decades, the basis of an ecosystem approach to fisheries (EAF) has been elaborated and prioritized in terms of both scientific and management. Key frameworks, plans, and commitments have paved the way towards implementation of an EAF around the world. The 1995 FAO Code of Conduct for Responsible Fisheries (Garcia, 2000) provided a reference framework for incorporating ecosystem considerations into sustainable fisheries management, and the 2001 Reykjavík Declaration (FAO, 2002) and the 2002 UN Sustainable Fisheries Resolution committed nations to implementing an EAF, individually and collectively, with the aim of reinforcing responsible and sustainable fisheries in the marine ecosystem. The concepts of EAF are no longer new, but practical implementation of an EAF remains a challenge and is yet to be achieved.

The development and the monitoring of ecological and socio-economic indicators play a prominent role for supporting the implementation of EAF by assessing ecosystem status, the impacts of human activities, the effectiveness of management measures, and communication of complex fishing impacts to a non-specialist audience (Cury and Christensen, 2005; Jennings, 2005; Rice and Rochet, 2005). Created in 2005 under the auspices of EurOceans, the IndiSeas scientific Working Group (WG) uses ecological indicators to analyse the impacts of fishing on the structure and functioning of marine ecosystems and to communicate these complex ecosystem effects into a digestible form beyond the scientific sphere, e.g. to resource managers and the general public. The WG attempted a comparative analysis and evaluation of the ecological status of the world's marine ecosystems, with specific focus on the ecosystem effects of fishing (Shin and Shannon, 2010). First, a suite of ecological indicators common to all the ecosystems studied was selected, then the data required for these indicators for 19 ecosystems spread over the Atlantic and Pacific Oceans (Table 1), involving 32 countries, were collected. The work reported in this suite of papers and presented on the IndiSeas website was undertaken as a meta-analysis by the IndiSeas WG (Shin and Shannon, 2010).

The number of ecosystem indicators under consideration has burgeoned over the past decade (Cury and Christensen, 2005; Piet *et al*., 2008), threatening to confuse rather than to augment traditional single-species assessments and management approaches. The challenge of the IndiSeas WG was not to develop new indicators, but rather to use specific selection criteria (Rochet and Trenkel, 2003; Rice and Rochet, 2005) to help select the most representative and practically achievable and meaningful set from those previously proposed. The overall aim was to assess the ecological status of exploited marine ecosystems and the changes they are undergoing. This implied that potential data to calculate the indicators needed to be readily available across a wide spread of marine ecosystems. An important characteristic of the indicators selected is that they are mostly survey-based (collection of scientific data in the field by boat-based surveys independent of the fishery), in contrast to most other comparative studies of fished ecosystems, for which meta-analyses have been based largely on model-derived or catch-based indicators (Alder and Pauly, 2008; Halpern *et al*., 2008; Pitcher *et al*., 2009). One of the consequences of this is that the IndiSeas WG relied strongly on multi-institutional collaboration, allowing sharing of scientific data and, above all, scientific diagnoses based on local expertise in each ecosystem investigated. Therefore, the global comparative approach kept a good track of the data underlying the indicators, and local scientific knowledge of the functioning of ecosystems is accounted for in the final diagnosis.

We here provide the background material to the entire suite of papers presented. These papers successively propose an evaluation of the states and trends of marine ecosystems regarding fishing effects, classifications, and ranking of ecosystems using a set of complementary or alternative methods (Blanchard *et al*., 2010; Bundy *et al*., 2010; Coll *et al*., 2010; Jouffre *et al*., 2010; Link *et al*., 2010; Shannon *et al*., 2010; Shin *et al*., 2010). In this manuscript, we describe the process through which the WG set the framework for subsequent comparative analyses of exploited marine ecosystems. The background framework includes: (i) the adoption of criteria for selecting a list of useful indicators which would be common to a wide range of ecosystems, (ii) the establishment of a common protocol for calculating and standardizing the selected indicators, and (iii) the adoption of a simple graphic representation of ecosystem states and trends.

A comparative approach across world marine ecosystems

Assessing the status of fish stocks can be difficult and prone to uncertainty; the task of assessing an ecosystem is far more challenging because there are no, or few, reference points at an ecosystem level (Jennings and Dulvy, 2005; Greenstreet and Rogers, 2006; Shin *et al*., 2010), only incomplete datasets are available, and ecosystems are non-linear systems that can be difficult to model and to predict. We are still learning how whole ecosystems respond to the effects of fishing in combination with environmental effects. The status of an ecosystem is the result of multiple factors and needs to be assessed in this light. One way to help facilitate ecosystem assessments and the implementation of an EAF is through comparative ecosystem studies, either focusing on single species (e.g. Brander, 1994; Drinkwater, 2005), whole ecosystems (e.g. Shannon and Jarre-Teichmann, 1999; Hunt and Megrey, 2005; Moloney *et al*., 2005; Bundy *et al*., 2009), or ecosystem indicators (Pauly *et al*., 1998; Alder and Pauly, 2008; Coll *et al*., 2008a, b; Pitcher *et al*., 2009; Shannon *et al*., 2009). Comparisons of similar ecosystems [upwelling and comparable ecosystems are dealt with by Shannon *et al*. (2010), and rankings according to ecosystem type are examined by Coll *et al*. (2010)] can serve as *ad hoc* replicates, to some extent mimicking an experimental set-up where common, unique, and basic features, as well as important responses to fishing, can be explored. At the same time, comparing ecosystems with contrasted exploitation and environmental conditions (as covered in the set of 19 ecosystems examined by Blanchard *et al*., 2010; Bundy *et al*., 2010; Coll *et al*., 2010; Link *et al*., 2010; Shin *et al*., 2010) can help determine the relative status of each ecosystem. With the difficulty in establishing baseline levels and reference points for most ecosystem indicators (but see Shin *et al*., 2010), a comparative approach across ecosystems may provide a range of indicator values against which each ecosystem can be assessed in relative terms. The more ecosystems included in the comparative analyses covering wide ranges of indicator values, the more significant the comparative analysis would be. Such comparative analyses provide an opportunity for taking a broader ecosystem perspective. By learning from mistakes made in degraded ecosystems and from early

Table 1. Ecosystems considered in the comparative approach, and the corresponding FAO fishing zones (http://www.fao.org/fi/website/FISearch.do?dom=area).

Coastal ecosystem	Geographic area	Type of ecosystem	Surrounding countries	Large marine ecosystem	FAO fishing zones[a]
Adriatic Sea (north-central)	Central Mediterranean	Temperate	Italy, Slovenia, Croatia, Bosnia-Herzegovina, Montenegro	Mediterranean	MFA: 37, Div: 2.1
Baltic Sea (central)	NE Atlantic	Brackish temperate	Germany, Estonia, Sweden, Poland, Russia, Lithuania, Latvia, Finland, Denmark	Baltic Sea	MFA: 27, Div: IIId 25 – 29
Barents Sea	NE Atlantic	High latitude	Norway	Barents Sea	MFA: 27, Div: I, IIb
Bay of Biscay	NE Atlantic	Temperate	France	Iberian coastal	MFA: 27, Div: VIIIa,b
Benguela (southern)	SE Atlantic	Upwelling	South Africa	Benguela Current	MFA: 47, Div: 1.6, 2.1
Bering Sea, Aleutian Islands	NE Pacific	High latitude	Alaska, USA	East Bering Sea	MFA: 67
Canada coast (West)	NE Pacific	Seasonal upwelling	Canada	Gulf of Alaska	MFA 67
Catalan Sea (southern)	NW Mediterranean	Temperate	Spain	Mediterranean	MFA: 37, Div: 1.1
Guinean EEZ	East-central Atlantic	Upwelling	Guinea	Guinea Current	MFA: 34, Div: 3.13
Humboldt (northern)	SE Pacific	Upwelling	Peru	Humboldt Current	MFA: 87, Div: 1.1, 1.2
Humboldt (southern)	SE Pacific	Upwelling	Chile	Humboldt Current	MFA: 87, Div: 2.14 – 2.27
Irish Sea	NE Atlantic	Temperate	Ireland, UK	Celtic – Biscay Shelf	MFA: 27, Div: VIIa
Mauritanian EEZ	East-central Atlantic	Upwelling	Mauritania	Canary Current	MFA: 34, Div: 3.12
Morocco (Sahara coastal)	East-central Atlantic	Upwelling	Morocco	Canary Current	MFA: 34, Div: 1.3
North Sea	NE Atlantic	Temperate	UK, Norway, Denmark, Germany, Netherlands, Belgium	North Sea	MFA: 27, Div: IVa,b,c
Portuguese EEZ	NE Atlantic	Upwelling	Portugal	Iberian coastal	MFA: 27, Div: IXa
Scotian Shelf (eastern)	NW Atlantic	Temperate	Canada	Scotian Shelf	MFA: 21, Div: 4V, 4W
Senegalese EEZ	East-central Atlantic	Upwelling	Senegal	Canary Current	MFA: 34, Div: 3.12
US coast (Northeast)	NW Atlantic	Temperate	United States	NEUS continental shelf	MFA: 21, mainly Div: 5Y, 5Zu, 5Zw, 6A, 6B, 6C

[a]MFA, FAO major fishing area; Div, FAO Division.

warning signals (declining indicators, in our case) picked up through comparative studies, fishery managers can be alerted to potential problems. Moreover, by comparing exploitation effects in ecosystems, generalizations can be drawn that may be important in improving the assessment of exploited marine ecosystems.

A comparative approach can also help in selecting robust ecological indicators that would be meaningful and measurable over a set of diverse and contrasted situations, as well as in specifying their conditions of use. The comparative approach between ecosystems and its communication to the general public is also aimed at creating an incentive for politicians to consider their management options, taking responsibility for the ecological quality of world marine ecosystems. The capability of indicators to be understood by the general public is important in implementing ecosystem-based management for analysis and discussion by society as a whole, ensuring stakeholder buy-in.

There are no objective criteria to determine the spatial extent of an ecosystem. For the purposes of this study, experts from each ecosystem were consulted and the best compromise in delineating an ecosystem adopted based on the information available (the extent of the area) and the representivity of the underlying physical, biological, and anthropogenic processes. Regional units such as large marine ecosystems (LMEs) were not always suited to our purpose because they are delineated according to bathymetry, hydrography, productivity, and trophically dependent populations (Sherman, 1991), not necessarily on historical levels of fishing activity. It must be kept in mind that the objective of the IndiSeas project was to study the ecosystem responses to fishing pressure. Whenever possible, it was proposed that ecosystems be considered as zones wherein fishing level can be regarded as relatively homogeneous. Therefore, the IndiSeas WG sometimes relied on existing management areas (e.g. exclusive economic zones, EEZs). In addition, LMEs are relatively large regions over which it would have been difficult to obtain a wide coverage of consistent and inter-calibrated scientific surveys on which many of the estimates of the selected ecological indicators are based.

For our work, 19 ecosystems that displayed disparity in terms of data sampling and quality and length of time-series (Appendix Table A1) were included. They include temperate, tropical, upwelling, brackish, and high-latitude ecosystems (Figure 1), span different socio-economic realities, vary in ecosystem structure and environmental forcing, include a range of exploitation histories

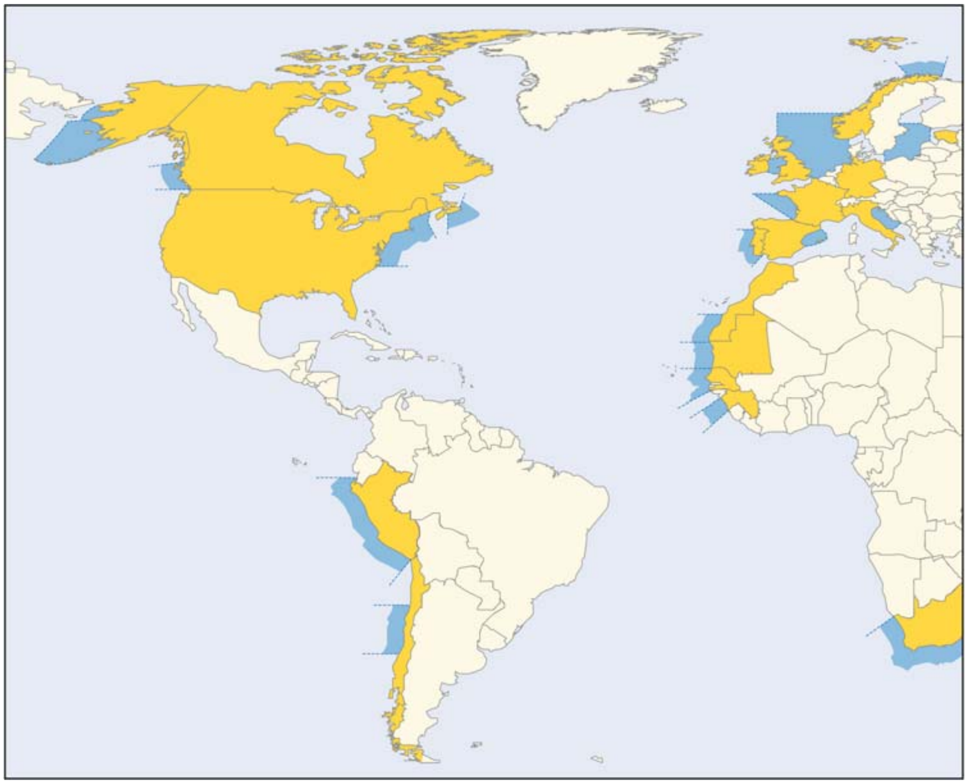

Figure 1. Marine ecosystems considered by the IndiSeas WG. Blue, the marine ecosystem; yellow, the countries participating in the analyses.

(Appendix), and span different socio-economic realities. The work represents the start of a global comparative analysis and diagnosis of ecosystem status, the results of which will be available to the scientific and general public through the dashboard developed for the IndiSeas website (Shin and Shannon, 2010; www.indiseas .org). Detailed descriptions of each ecosystem are available on that IndiSeas website, and summaries of each are provided in the Appendix. The names and affiliations of the experts who participated in the first phase of IndiSeas are listed in Appendix Table A2.

Selection criteria

For the selection of a common list of indicators, the WG adopted a step-by-step process, as inspired by Rice and Rochet (2005), which started by defining the objectives of the group and the subsequent requirements that the indicators needed to fulfil. The aim was to evaluate and compare the ecological status of exploited marine ecosystems, with the potential for worldwide coverage, and to communicate the results of the comparative approach to a large audience. Among the multitude of indicators that can characterize the state and trends of a marine ecosystem, we chose to focus on ecological ones that can help track the effects of fishing at ecosystem and community levels, to be used in the framework of a comparative approach, and which are easily communicated to the general public. Based on this general framework, the second step was to develop a list of candidate indicators, based on a review of the literature, with a clear intent to build on the work already achieved by the international SCOR/IOC WG 119 on "Quantitative Ecosystem Indicators" (2001–2004). In their review of existing ecosystem indicators, three categories of ecological indicator were distinguished by the members of the SCOR WG (Cury and Christensen, 2005), namely size-based indicators (Shin *et al*., 2005), species- or life-history-based indicators (Jennings *et al*., 1999; Greenstreet and Rogers, 2006), and trophodynamic indicators (Cury *et al*., 2005). The list of candidate indicators developed by the IndiSeas WG attempts to reflect this diversity of ecological indicators (Table 2). Pressure indicators (Degnbol and Jarre, 2004; Jennings, 2005), and simple biomass-related indicators were also included in the list of potential, ecological indicators.

The third step was to determine screening criteria to select a tractable set of indicators among the many existing ecological indicators, notably within the framework of a comparative approach across ecosystems. Again, the IndiSeas WG based its work on the results of SCOR WG 119, adopting and adapting some of the practical criteria outlined by Rice and Rochet (2005) in selecting ecosystem indicators:

(i) *Theoretical basis/ecological significance*: indicators should reflect well-defined ecological processes occurring under fishing pressure. The indicators should rely on strong scientific and theoretical knowledge of the links between communities and fishing pressure.

(ii) *Sensitivity*: the indicator should be able to track ecosystem changes attributable to fishing. Trends in the indicator need to be highly correlated with the trends in fishing pressure.

(iii) *Measurability*: this criterion encompasses what Rice and Rochet (2005) label as cost, measurement, and historical data criteria. Succinctly, the indicators need to be measurable or estimated on a routine basis, and data time-series must be available.

Table 2. Initial list of candidate indicators evaluated against four screening selection criteria (ecological significance, sensitivity, measurability, awareness of the public), crosses (x) meaning that the indicator satisfies the selection criterion, according to the expertise of the IndiSeas WG.

Indicator	Ecological significance	Sensitivity	Measurability	Public awareness	Management objective[a]
Size-based indicators (Link, 2005; Rochet and Rice, 2005; Shin *et al.*, 2005)					
Mean length/weight in community	x	x	x	x	EF
Maximum length in community	x	x		x	
Mean maximum length in community	x	x		x	
Slope of size spectrum	x	x			
Slope of diversity size spectrum	x				
Proportion of large fish	x	x		x	
Proportion of large species	x	x	x	x	CB
Trophodynamic indicators (Cury *et al.*, 2005; Fulton *et al.*, 2005; Link, 2005; Pauly *et al.*, 2000)					
TL landings	x	x	x	x	EF
TL community	x	x	x	x	EF
Fishing-in-Balance index	x	x			
Proportion of predatory fish	x	x	x	x	CB
Pelagic to demersal fish biomass ratio	x	x	x		
Piscivorous to zooplanktivorous fish biomass ratio	x	x	x		
Species-based indicators (Degnbol and Jarre, 2004; Fulton *et al.*, 2004; Link, 2005; Rochet and Rice, 2005; Yemane *et al.*, 2005)					
Species richness	x			x	
Shannon and Hill's index of diversity	x		x		
K-dominance, ABC curves, *W*-statistic	x	x	x		
Ratio of endangered to unendangered species	x	x		x	
Ratio of target to non-target species	x		x	x	
Proportion of sustainably or under-/moderately exploited stocks	x	x	x	x	CB
Mean lifespan	x	x	x	x	SR
Pressure indicators (Degnbol and Jarre, 2004; Fréon *et al.*, 2005; Fulton *et al.*, 2005)					
Overall fishing mortality rate	x	x	x	x	RP
Exploited fraction of ecosystem surface	x		x	x	
Mean distance of catches from the coast	x				
Catch rate by community	x	x			
Discard rate	x			x	
Biomass-related indicators (Blanchard and Boucher, 2001; Fulton *et al.*, 2005; Link, 2005; Rochet *et al.*, 2005)					
Total community biomass	x	x	x	x	RP
Coefficient of variation in biomass	x	x	x	x	SR

[a]CB, conservation of biodiversity; SR, maintaining ecosystem stability and resistance to perturbation; EF, maintaining ecosystem structure and functioning; RP, maintaining resource potential.

(iv) *General public awareness*: the meaning of the indicator and its link with fishing needs to be widely and intuitively understood. Meeting this criterion would allow indicators to meet the "concreteness" criterion (Rice and Rochet, 2005), requiring that we avoid abstract ecological features. Hence, the concreteness criterion was not addressed separately.

Criteria (i) and (ii) were taken as a basic requirement that had to be fulfilled by the final set of indicators, but were not really determinant in prioritizing indicators. Clearly, indicators should be sensitive to fishing, but the extent to which some indicators are more-or-less sensitive relative to others is not systematically documented. There are some empirical analyses using multiple indicators that can test the relative performance of indicators in detecting changes in fishing, but they tend to be specific to the ecosystem studied (Trenkel and Rochet, 2003; Greenstreet and Rogers, 2006) and, as shown by Travers *et al.* (2006), the relative sensitivity of indicators also depends on the fishing scenario. It is common sense to anticipate that indicators such as species richness or the exploited fraction of the ecosystem surface (Table 2) will not be highly sensitive to fishing or will rapidly plateau, but for most candidate indicators, generalities cannot easily be drawn in the context of a worldwide comparative approach. Modelling approaches can help in defining the range of sensitivities of indicators (Travers *et al.*, 2006) and testing the robustness of indicators (Fulton *et al.*, 2005), but more systematic simulations should be undertaken using multiple indicators in different fishing contexts and in different types of ecosystems to be able to draw generalities.

The most constraining criterion in the comparative framework was that of data availability in the different ecosystems (from direct observations or from model output). The concise assemblage of indicators needed to be comparable across ecosystems, and estimation of the indicators not too costly. In other words, they needed to be estimated easily and gathered for each ecosystem considered. Application of the list of candidate indicators in the 19 ecosystems included in the comparative approach showed that individual size data were not recorded systematically, precluding estimation of many of the size-based indicators (Table 2). Pressure indicators at the level of the community were also difficult to estimate. Another important concern to the IndiSeas WG was that of the awareness of the general public of the meaning, i.e. the information communicated, of each indicator. The objective was to retain the indicators for which there could be a direct and transparent link between the ecological meaning of the indicator and its public perception. For example, among potential size-based indicators, the slope of the size spectrum was considered to be a second-order indicator, derived from statistical models of individual data (Travers *et al*., 2006), which would be difficult to communicate to the general public.

Of course, fishing is not the only driver potentially influencing marine ecosystems. The effects of changes in fishing strategy and in climate forcing, for example, can be mutual and intricate (Frank *et al*., 2006; Lees *et al*., 2006), and this is reflected by the response of ecological indicators of fish communities (Blanchard *et al*., 2005; Shin *et al*., 2005). Apart from pressure indicators (Jennings, 2005), there are few or no ecological indicators that can be considered exclusive to fishing. Very few empirical or modelling studies can discriminate the utility of indicators in terms of their specificity to fishing impacts, so the specificity criterion was not helpful and hence not used for selecting the final set of indicators. The consequence is that, as in any other indicator-based study, care had been taken in interpreting the indicators, and their situation had to be contextualized. Therefore, the impact of confounding factors needed to be taken into account, and this aspect is broadly discussed in the suite of papers presented here (Coll *et al*., 2010; Link *et al*., 2010; Shannon *et al*., 2010). Inputs from scientific experts from the various ecosystems helped in disentangling fishing impacts from the impacts of other drivers.

Likewise, the responsiveness criterion (the rapidity with which an indicator responds to a change in the driver of interest, i.e. fishing) was not used to select the set of indicators because, again, current knowledge of the ecological processes could not help in discriminating the utility of indicators based on it. As they are more integrated than population indicators, community and ecosystem indicator responses to changes are smoothed, and generally slower. The statistical portfolio effect is just one aspect that can explain the lack of immediate response (Shin and Cury, 2001). Nicholson and Jennings (2004) showed that the statistical power of community indicators in detecting trends is low for time-series of $<$10 years. This threshold was therefore applied for the analyses of trends in all the indicators selected by the IndiSeas WG (Blanchard *et al*., 2010; Bundy *et al*., 2010; Coll *et al*., 2010; Link *et al*., 2010; Shannon *et al*., 2010).

In addition to these practical selection criteria, we set four ecological ecosystem-level attributes (Table 2) to help define what Rapport *et al*. (1998) refer to as ecosystem health, which had to be reflected by the final set of indicators and which can be linked to management strategic priorities (Murawski, 2000): (i) conservation of biodiversity (CB), which appears to be a legal obligation in many international conventions (Greenstreet, 2008); (ii) maintenance of ecosystem stability and resistance to perturbation (SR), which refers to the counteractive capacity described by Rapport *et al*. (1998); (iii) maintenance of ecosystem structure and functioning (EF); and (iv) maintenance of resource potential (RP) in terms of production capacity, also referred to as vigour by Rapport *et al*. (1998). For each indicator, we identified the management objective to which it was most clearly associated, although some indicators could easily be used for addressing several management objectives in parallel. The four specific objectives are indeed inter-linked, and this becomes clear when there is excessive exploitation in an ecosystem. Overfishing may reduce biodiversity, influencing fish populations at various levels of organization (Greenstreet and Rogers, 2006). This can be exhibited in multiple ways, e.g. in a reduction in size structure (mean length) and/or altered genetic variability within species, impacting the ecosystem stability and its resistance to perturbations. When fishing alters the relative abundance patterns of species, predator–prey relationships may be disrupted and effects may cascade through the foodweb, altering ecosystem structure and function as well as resource potential. In this case, the advocated management action of decreasing overall fishing mortality should eventually lead to recovery of the ecosystem, although this may take decades; alternatively, exploitation may drive an ecosystem to a changed state, resulting in irreversible effects of exploitation, such as appears to have happened on the eastern Scotian Shelf (Bundy and Fanning, 2005). In some highly dynamic ecosystems, such as where there is upwelling, climate changes can give rise to alternating dominance by species. It is more difficult to interpret whether the selected indicators address the four management objectives in such cases, but identifying regime shifts does help in understanding and interpreting state and trend indicators for those ecosystems.

The indicators

Justification of the final selection

To summarize, the selection of the final set of indicators by the IndiSeas WG followed three simple rules: the selected indicators had to fulfil the four main criteria listed (ecological significance, sensitivity, measurability, and general public awareness), there had to be at least one indicator per category (size-based, species-based, trophodynamic, pressure, biomass-related), and at least one indicator per management objective. To achieve a balance among the management objectives, and in light of the remaining ten indicators that fulfilled the four selection criteria (Table 2), we decided to retain two indicators per management objective. Trophic level (TL) of the landings was preferred to TL of the community for two reasons: it reflects only the recruited phase to which the TL estimates per species best correspond, and it provides a complementary view to other indicators based mainly on survey data. Further, the proportion of predators was preferred to the proportion of large species because the latter indicator necessitates choosing an arbitrary threshold for L_∞ at which a species is considered to be large.

To facilitate communication, each indicator selected was given a headline label (Table 3), and indicators were all formulated positively, so that a low value of an indicator reflected strong impacts of fishing, and a higher value suggested weaker fishing impacts. Similarly, an increase in an indicator meant an improving state, whereas a decrease was assumed to reflect deterioration of an ecosystem as a result of fishing.

The eight indicators listed in Table 3 were selected based on the above criteria and are proposed from now on for diagnosing the status of a fished marine ecosystem. Six of them were used to measure the state (S) of the ecosystem, and six (of which two differ from state indicators) were used to measure trends (T) over time (Table 3). Data for the indicators were derived primarily from fisheries-independent (survey) and fisheries-dependent (commercial catch) data, with auxiliary information used where indicated (such as ecosystem models). For example, there is no common fisheries survey in the Baltic Sea (separate surveys are

Table 3. Summary of ecological indicators selected by the IndiSeas WG and the corresponding management objectives.

Indicators	Headline label	Used for State or Trend	Management objective[a]
Mean length	Fish size	S, T	EF
TL of landings	TL	S, T	EF
Proportion of under- and moderately exploited stocks	% healthy stocks	S	CB
Proportion of predatory fish	% predators	S, T	CB
Mean lifespan	Lifespan	S, T	SR
1/*CV* of total biomass	Biomass stability	S	SR
Total biomass of surveyed species	Biomass	T	RP
1/(landings/biomass)	Inverse fishing pressure	T	RP

[a]CB, conservation of biodiversity; SR, maintaining ecosystem stability and resistance to perturbation; EF, maintaining ecosystem structure and functioning; RP, maintaining resource potential.

directed at cod and pelagic planktivores, the time-series for which are relatively short), so the output of a multispecies virtual population analysis was used for the Baltic Sea (ICES, 2006a).

In some ecosystems, the data required to calculate the indicators selected have not yet been collected or are not readily available. However, the WG felt that it was important in those cases to set up sampling or modelling programmes to fill the gaps. Therefore, the list proposed by IndiSeas (Table 3) is not strictly the lowest common denominator of all the ecosystems represented. Some indicators were retained in the final list, although not available yet for all ecosystems, because it was believed that they could be estimated in future at relatively low cost.

Proposed set of indicators

"Total biomass of surveyed species" is generally observed to be a more stable indicator than species biomass (Sutcliffe *et al.*, 1977; Murawski *et al.*, 1991); as species are fished and their biomass is reduced, other species (competitors, prey) can increase in abundance and replace them in the foodweb. However, differences in habitat and prey specificities between species will determine the extent of species replacements and competitor responses. Yet, where exploitation is generalized through the foodweb with high levels of fishing mortality applied, the total biomass of fish communities is expected to decrease (Rochet *et al.*, 2005). Fishing mortality can be of two types, directly related to fishing and foodwebs effects induced by fishing. The former includes catches of target species and the unintended mortality of non-target and bycatch species, through discarding. Foodweb effects include mortality attributable to increased predation or competition propagated through the foodweb as a result of species changes caused by fishing (Bundy *et al.*, 2005). As a biomass decrease in a community would impair the productivity of dependent fisheries (Rochet and Trenkel, 2003), the term biomass is used here as a measure of resource potential, referring to the production capacity and the potential contribution of the ecosystem as an exploitable marine resource (Table 3). The biomass estimated from surveys is generally not absolute, but rather an index of fish density (owing to differences in catchability between the species surveyed), and therefore not comparable in absolute terms across ecosystems (for characterizing ecosystem states). Instead, biomass is useful as a relative indicator when considering changes in indicators over time (i.e. trends). Biomasses reported depend on the communities surveyed to obtain the indices, and it is often difficult to combine biomass indices from different types of survey.

"1/(landings/biomass)" measures the inverse level of exploitation or total fishing pressure on an ecosystem. This indicator is considered as a measure of resource potential (Table 3) because it reflects the part of the community production dedicated to fishing. It is more commonly expressed as landings/biomass (as a proxy for exploitation rate), but it was inverted here so that it should decrease under increasing fishing pressure, hence varying theoretically in the same direction as the other indicators in the selected suite. This indicator is the only one from the suite that can be considered as a potential pressure indicator (Jennings, 2005). Notwithstanding, care needs to be taken in interpreting trends in this indicator because variations in total biomass and catch are not only the result of fishing (see Table 5 of Bundy *et al.*, 2010). Further, it is influenced by changes over time in the selectivity of fishing gear and in the species targeted by fishing sectors, as well as by inconsistencies in reported catches (see Bundy *et al.*, 2010, for further discussion). Direct comparable pressure indicators were not available for this study, and there is not consensus within the scientific community as to how to estimate fishing pressure at an ecosystem level. Daan *et al.* (2005) proposed to estimate trends in the rate of community exploitation by averaging fishing mortality rates for individual species as means weighted by the average biomass of each species over the entire period studied. However, many ecosystems considered in the IndiSeas project encountered problems of data availability in providing this type of pressure indicator, so the simplest way to proceed in reconciling data availability and ecological meaning was to consider the ratio of landings over biomass. As for total biomass, this indicator is used only for comparing trends, because it is a derivative of biomass data, for which absolute estimates are generally not available.

"Mean length of fish in the community" allows tracking direct fishing effects on an ecosystem (Shin *et al.*, 2005). Size-based indicators generally crystallize a number of processes triggered by fishing: high-value target species are generally larger, fishing gears are size-selective and often designed to remove larger fish, older (and larger) fish in a population become fewer because cohorts accumulate the effects of fishing mortality through time, and large species are more vulnerable because their life-history traits are generally linked to lower potential rates of increase (Jennings *et al.*, 1998, 1999; Shin *et al.*, 2005). From a single-species perspective, the removal of larger fish, which are more fecund and produce more-viable eggs than smaller fish (Longhurst, 1999), compromises population productivity. From an ecosystem perspective, the removal of larger species changes the size structure of the community and potentially also ecosystem functioning. Recognizing that changes in species composition and fluctuations in recruitment can alter this indicator (Shin *et al.*, 2005), fish size generally decreases under fishing pressure. As many ecosystem processes are size-based and size provides information on the metabolic and trophic structure of marine ecosystems, this indicator is taken here as a measure of ecosystem structure and functioning (Table 3) and is used to measure the state and trend in the ecosystems.

"TL of landings" measures the weighted mean TL of species exploited by the fishery, representing the trophic position of the whole catch, and is expected to decrease in response to fishing, because fisheries tend to target species at higher TLs first (Pauly *et al*., 1998). Initially, catches increase as the foodweb is fished down and because lower TLs are ecologically less expensive (production is greater at lower TL and there is less loss of productivity by trophic transfer up the foodweb), catches may ultimately stabilize or decline (Pauly *et al*., 1998). Fishing can change the structure of marine foodwebs by reducing the mean TL and potentially also ecosystem functioning by shortening the length of food chains and releasing predation on low-trophic-level organisms. TL is considered to be a measure of ecosystem structure and functioning and is used to measure state and trend (Table 3). The TL of individual species is estimated either through modelling or dietary analysis, or taken from a global database such as Fishbase (www.fishbase.org). As has been shown by empirical or modelling studies (Jennings *et al*., 2002; Marzloff *et al*., 2009), TL can vary with fish age because fish are life-history-dependent omnivores. By considering TL of landings (vs. TL of the community), however, we focus on the recruited/adult stages of the populations to which the species TL estimates better correspond. In addition, by taking an average value per species (instead of considering individual values within each species), the principle here is to have an indicator of the species composition of the catch in terms of trophic positioning, rather than to track fishing effects on the TL of single species. Additionally, it would be totally unrealistic to try to collect individual data on TL over the set of ecosystems considered here. Again, therefore, the measurability criterion determined the selection of the indicator and the way it was calculated.

"Proportion of predatory fish" is a measure of the diversity of fish in the community and reflects the potential effects of fishing on the functioning of marine foodwebs. The resilience of predator species is particularly threatened by intense exploitation (Hutchings, 2000; Christensen *et al*., 2003; Myers and Worm, 2003), but their role in the ecosystem is essential because they act as dampeners of the whole foodweb (Sala, 2006), and their depletion can lead to trophic cascades (Frank *et al*., 2005, 2006; Daskalov *et al*., 2007). Restoring the declining abundance of predator functional groups should be a target of EAF implementation (Daskalov, 2008). The indicator is considered to be associated with the management objective CB (Table 3) and is used to measure state and trend. For its calculation here, predatory fish were considered to include all fish species surveyed that are piscivorous or feed on invertebrates >2 cm.

"Proportion of under- and moderately exploited stocks" represents the extent of success of fisheries management and is associated with the management objective CB (Table 3). Ideally in a precautionary world, all stocks should be at least moderately exploited to ensure sustained biodiversity and sustainable ecosystems. The FAO classification of stocks as underexploited, moderately, fully, or overexploited (http://www.fao.org/docrep/009/y5852e/Y5852E10.htm#tbl) was used to define these categories for the stocks in each ecosystem under consideration. Therefore, the indicator is used to compare the state of ecosystems. FAO data were used in most ecosystems to calculate it. Compared with other indicators in our list, the proportion of under- to moderately exploited stocks satisfies the criterion of exclusiveness to fishing pressure (Rice and Rochet, 2005), which makes it the least ambiguous to interpret with regard to other potential drivers, provided the method of estimation is reliable.

"Mean lifespan" is a proxy for the mean turnover rate of species and communities and is meant to reflect the stability of a system. It is therefore considered to be a measure of ecosystem stability and resistance to perturbations (Table 3), and is used to characterize both state and trend. Species with a short lifespan generally have a fast turnover rate and their dynamics will tend to be subject to environmental variability (Winemiller, 2005). In contrast, long-lived species tend to dampen ecosystem variability (Hsieh *et al*., 2006). The lifespan or longevity is considered here to be a fixed parameter per species, so the mean lifespan of a community will reflect the relative abundances of species with different turnover rates. Fishing influences the longevity of a species (the direct effect of fishing and genotype selection), but the purpose here was to track changes in species composition (assuming the same principle as for the mean TL of the catch).

"1/coefficient of variation (CV) of total biomass" is adopted as a measure of the stability of the ecosystem and is assumed to be affected by fishing (Blanchard and Boucher, 2001; Fulton *et al*., 2004; Hsieh *et al*., 2006). As with fishing pressure, it is expressed as an inverse, to conform with the directionality of the other indicators. Therefore, a low $1/CV$ indicates low biomass stability, and a low ecosystem stability and resistance to perturbations. As the indicator is measured over the past 10 years ($1/CV$ for 1996–2005), it is only used to measure state. At the population level, the rationale behind it is that exploited populations would experience increased variability in biomass as a consequence of fishing-induced truncation of the age structure, which reduces the capacity of populations to buffer environmental events (Hsieh *et al*., 2006). At the community level, fishing would tend to affect longer-lived, large-bodied species to a greater extent (Shin *et al*., 2005), and they generally exhibit greater variability in recruitment (Winemiller and Rose, 1992). Additionally, as total biomass decreases, the area occupied by the various stocks may decrease, the stocks may be more patchily distributed, or they may occupy the same area at a lesser density. All these cases will result in a more-variable survey index and hence an increased CV (and reduced biomass stability).

Calculation of indicators

Calculation of the eight selected ecological indicators is detailed in Table 4. To guide the calculations of indicators, groupings of the species considered (retained, surveyed, and predator species) were clearly defined along with their corresponding species parameters (Table S1). Most indicators relied on fisheries-independent survey data. The exceptions were TL of the landings and the proportion of under- to moderately exploited stocks, which used catch data and the output of stock assessment or ecosystem models. The source of data for the calculation of indicators is diverse, including scientific surveys, records of commercial catches, stock assessment output, and estimates of species parameters such as lifespan and TL (Appendix Table A1). However, these data requirements appeared to be tractable in all the ecosystems we studied. Specifically, Jouffre *et al*. (2010) explore and discuss some of the issues and constraints encountered in estimating trawl-based indicators. In terms of survey data in general, it is critical to determine whether or not they should include prerecruits in the evaluation of indicators. For most population and ecosystem indicators, interpretation of their trends will differ according to the life stages considered, so the diagnosis may

Table 4. Minimal list of indicators for establishing the dashboard [*L*, length (cm); *i*, individual; *s*, species; *N*, abundance; *B*, biomass; *Y*, catch; TL, trophic level).

Indicator	**Calculation, units**	**Comments to guide calculation of indicators**
Mean length	$\bar{L} = \sum_i L_i/N$ (cm)	*Data*: all surveyed species[a] *Question: in places where there is no data for length, what about weight?* Weights are converted to lengths using weight–length relationships Reason for choosing length—more meaningful to public; length is less directly affected by environmental change
TL landings	$\overline{TL}_{\mathrm{land}} = \sum_s (TL_s Y_s)/Y$	*Data*: Fixed non-integer TL per species. All retained species[b]. Can be calculated from Ecopath model or diet data *Question: if there is no Ecopath model implemented nor diet data available, can this indicator be calculated?* As a stopgap, the estimates of TL in Fishbase (www.fishbase.org) are used
Proportion of under- and moderately exploited stocks	Number (under- + moderately exploited stocks)/total number of stocks considered	*Method*: Three methods were tested: (i) using only local expertise on a list of assessed stocks, (ii) using only FAO database (stock status and number of assessed stocks), (iii) using FAO stocks list but also local expertise to refine FAO assessments when possible The first method was biased because in some cases the number of assessed stocks was too low compared with the number of stocks that are actually exploited The second method was not always satisfying because FAO regions are too large compared with the ecosystems considered by the WG: the list of stocks was not always adapted and stock status not necessarily the same over the whole FAO region (e.g. stocks off Namibia and South Africa) The third method was adopted according to the following step-by-step procedure: • listing the stocks that are referenced by FAO in the area of concern (http://www.fao.org/docrep/009/y5852e/Y5852E10.htm#tbl) • cutting this FAO list according to what is effectively retained in the ecosystem (=total number of stocks considered) • adding local expert knowledge to refine the FAO classification of stock status and to fill the gaps, providing sources (WG reports, published literature, pers. comm.) The advantage of the above method is adoption of the same reference list of major world stocks that was already established by the FAO
Proportion of predatory fish	Proportion of predatory fish = biomass of predatory fish surveyed/biomass surveyed. Biomass surveyed = biomass (demersal fish + pelagic fish + commercially important invertebrates)	*Question*: *are invertebrate species to be included in the predator pool?* No, see definition of "predatory fish species"[c]. As such, this indicator can reflect a potential decrease in demersal stocks, and a parallel increase in forage or invertebrate species

Mean lifespan	$\sum_S(\mathrm{age}_{\max,s}\, B_S)/\sum_S B_S$ (years)	*Meaning*: Proxy for turnover rate. Conveys the idea that fishing favours the emergence of species with a short lifespan. Fishing may affect the longevity of a given species (phenotypic plasticity and genotype selection), but the purpose here is not to track those effects at the species level, but rather to track changes in species composition *Data*: Calculated for surveyed species[a]. Fixed longevity for each species. Lifespan may vary under fishing pressure, so we conventionally adopted the maximum longevity observed for each species
1/*CV* of total biomass	Mean(total biomass for the past 10 years)/s.d.(total biomass for the past 10 years)	*Data*: Biomass of all surveyed species[a]
Total biomass of surveyed species	*B* (t)	*Data*: All surveyed species[a]. Specific surveys conducted for sampling eggs, larvae, and juveniles should not be considered. This biomass index is used only for trends, so absolute biomass estimates are not needed *Question: do different surveys have to be combined (demersal trawl, pelagic acoustic . . .)?* In some cases, considering only demersal trawl surveys provides an adequate estimate of the biomass of demersal/pelagic fish and commercially important invertebrates. However, in some systems (such as upwelling ones), small pelagic fish are not adequately sampled in demersal trawl surveys, so dedicated small pelagic surveys are carried out. In those cases, local experts decide on appropriate methods of combining different surveys to provide a single total biomass index for the ecosystem
1/(landings/biomass)	B/Y retained species[b]	*Meaning*: Indicates global fishing pressure at the community level *Data*: Use total landings and biomass of retained species[b] Used for trends, so biomass indices can be used (but must be consistent across species and over the time-series)

[a]Surveyed species: these are species sampled by researchers during routine surveys (as opposed to species sampled in catches by fishing vessels) and should include 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 abalone and mussels, mammalian and avian top predators, and turtles, should be excluded. Surveyed species are those considered by default in the calculation of all survey-based indicators.

[b]Retained species (landed): these are species caught in fishing operations, although not necessarily targeted by a fishery (i.e. including bycatch species), and which are retained because they are of commercial interest, i.e. not discarded once caught, although this does not imply that sometimes certain size classes of those species may not be discarded. A non-retained species is considered to be one that would never be retained for consumptive purposes. Intertidal and subtidal crustaceans and molluscs such as abalone and mussels are excluded. Retained species are those that are considered by default in the calculation of all catch-based indicators.

[c]Predatory fish species: predatory fish are considered to be all surveyed fish species that are not largely planktivorous (i.e. phytoplankton and zooplankton feeders should be 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 should not be classified as predatory fish.

sometimes be biased. For example, when mean size decreases, it can be because of decreasing numbers of large fish, better recruitment, or both. Because our aim is to measure fishing effects rather than to capture the environmental variability reflected in variable recruitment success, we focused on recruits or adult fish. As a consequence, specific surveys conducted for sampling eggs, larvae and juveniles should not be considered in calculating indicators.

Among the indicators, three relied on the calculation of averages: mean length, mean lifespan, and mean TL of the landings. According to the way the averages are calculated, the focus is on different aspects of ecosystem structure. There are two ways to calculate averages at the ecosystem level: by considering all data available or by using some form of stratification and weighting (e.g. by species or functional group). For example, to calculate the mean size of fish at an ecosystem level, all individual sizes in survey samples need to be considered, as we did. In contrast, to calculate mean lifespan or mean TL of the landings, a fixed value per species was considered (e.g. the TL or the lifespan of a species), then the relative biomasses of the species were used to weight the average indicator at the ecosystem level. These two last indicators are therefore meant to reflect a change in species composition, not the fishing effects at the population level (decrease in TL or a decrease in the observed lifespan). In contrast, a decrease in the mean length in the community will be due to two confounding effects: the decrease in the mean length of individual species, and/or the decrease in the proportion of large species. Adopting two ways of calculating averages can provide complementary views on the changes in ecosystem structure (Travers *et al.*, 2006).

Synthesis and graphic representation of indicators

Images help us understand complex patterns (Boulding, 1956; Massironi, 2002; Jentoft *et al.*, 2008), and also to describe, analyse, and synthesize information (Cleveland and McGill, 1985). As such, they are ideal tools for conveying and synthesizing the information from a suite of ecosystem indicators such as those proposed in this IndiSeas project. We therefore developed a generic dashboard to present the ecosystem indicators describing the state of ecosystems and the trends within them, using pie diagrams and simple bar plots. The advantage of such representation lies in providing a multivariate view of the ecosystem, which is easily comparable across systems.

Pie diagrams (used in the IndiSeas suite of papers by Shannon *et al.*, 2010, and Shin *et al.*, 2010; www.indiseas.org) were used to present the results of the state analysis, where state indicator values (Table 3) were averaged over the years 2003–2005 to represent the current state of the ecosystem. Each pie corresponds to a selected indicator. On that axis, the indicator is scaled between a minimum value (centre of the diagram) and a maximum value, values constant across all ecosystems considered in the dashboard and determined by the minimum and maximum values observed in the set of ecosystems. The boundaries are not intended to represent optimum or target values, but are merely used to scale the indicators for graphic representation and comparative purposes. This approach highlights the importance of having an inclusive set of ecosystem case studies, so that the sizes of the respective pie slices reflect how the indicators compare relative to others across a broad cross section of ecosystem types.

Short- to medium-term trends were calculated over a 10-year period, 1996–2005, or over 25 years, 1980–2005, for the suite of six standardized trend indicators (Table 3). Bar plots were used to represent the trends that were significant (used here by Blanchard *et al.*, 2010, and Shannon *et al.*, 2010), positive and green in their figures indicating an increase and negative and red a decrease. Solid bars indicate statistically significant trends, pale bars indicate non-significant trends.

Conclusions

From the selection phase of ecological indicators within the IndiSeas project, and from the extensive statistical testing conducted on the 19 participating ecosystems, it soon became apparent that combining and comparing the sets of indicators (see specifically Bundy *et al.*, 2010; Coll *et al.*, 2010; Shin *et al.*, 2010) can be helpful in establishing a diagnosis of the status of exploited ecosystems. Some general lessons regarding indicators for advancing an EAF have been learned during the project (Shin and Shannon, 2010). This stems from the potential benefits of a comparative approach in understanding driving mechanisms of exploited marine ecosystems. Comparative analysis of ecosystem attributes (biomass, size, etc.) as functions of drivers (fishing pressure, and local/regional environmental drivers) showed insightful similarities and differences among ecosystems in terms of potential drivers (Coll *et al.*, 2010; Link *et al.*, 2010; Shannon *et al.*, 2010). We note a few of the main advantages here.

First, the set of indicators used for evaluating ecosystem states (Shin *et al.*, 2010) is slightly different from that used for trend analyses (Blanchard *et al.*, 2010; Bundy *et al.*, 2010), but in combination, both assessments form a useful means of assessing the ecological impacts of fishing (Coll *et al.*, 2010; Shannon *et al.*, 2010). Additionally, monitoring indicators that measure complementary and sometimes similar characteristics, and which combine survey and catch data, such as mean length of the community and mean TL of the landings, is encouraged and has proven useful in consolidating the diagnosis of fished marine ecosystems. For monitoring exploited populations and ecosystems, ecological indicators can be calculated from survey data and from catch data. Both sources of information can be complementary, because they do not necessarily include the same components: population indicators based on survey data may include broader age/size classes and provide information on most species, whereas catch data may provide information on the part of the population that is recruited to the fishery and important insight on other species taken as bycatch, which can be used in calculating indicators at the community/ecosystem level. Further, survey data usually have wide spatial coverage with limited temporal resolution, whereas catch data provide the benefit of greater temporal resolution because fishing activities are carried out over longer periods in any year. The combination of ecological indicators selected by IndiSeas suitably reflected different dynamics and tracked processes that cover different management goals and which may respond differently to fishing (e.g. mean length and biomass stability), and together provided a complementary means of assessing marine ecosystem change (Bundy *et al.*, 2010; Coll *et al.*, 2010) and state (Coll *et al.*, 2010; Shin *et al.*, 2010). In addition, the project has demonstrated that simple, often available, indicators can provide a good perspective of ecosystem status and the impacts of fishing, compared with what is known from other types of assessment (see Bundy *et al.*, 2010, and Shannon *et al.*, 2010, for further discussion). This is important because the socio-economics in areas where fishing activities develop differs significantly around the globe, and in many countries, insufficient data are available for complex and exhaustive analyses. Using simple yet rigorous, scientifically sound indicators through an ecosystem approach is a promising way to achieve real evaluation of marine ecosystem status as a result of fishing activities.

Another key lesson is that the interpretation of results by scientific experts representing each ecosystem is critical to correct interpretation of state and trend indicators and in disentangling the effect of other potential ecosystem drivers such as the environment or other anthropogenic impacts. The WG consisted of local experts (Appendix Table A2) responsible for collating and calculating the indicators, and who provided the necessary local perspectives and insights for interpreting the results and how they compared across ecosystems. The many contributions of local experts to the global comparative analyses were essential in the work of the IndiSeas WG, and experts from other ecosystems are encouraged to join any future IndiSeas initiative to expand the representivity of the set of ecosystems examined for fishing effects using ecological indicators. In future, effort will be made to expand the existing IndiSeas framework to incorporate indicators of conservation/biodiversity and socio-economic indicators to provide a broader indicator basis for EAF in its fuller sense (Shin and Shannon, 2010). In addition, it is planned that indicators summarizing environmental drivers, impacts, and changes will be explored in an attempt to define a generic or at least ecosystem type-specific set of indicators to capture natural drivers of ecosystem change. This will help to address the problems experienced in systems in which environmental drivers are strong, such as upwelling systems, and will combine multiple drivers of ecosystem change. Indicators of non-fishing-induced change, such as pollution, should also be identified where appropriate, and incorporated into the common framework.

Supplementary material

Supplementary material is available at ICESJMS online: for each of the 19 ecosystems considered in the IndiSeas suite of papers, a table is provided listing all species included in the calculation of the indicators and providing the species parameters used in calculating mean TL of the landings, mean lifespan, and the proportion of under- to moderately exploited stocks.

Acknowledgements

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 the IRD (Institut de Recherche pour le Développement). We thank Arnaud Bertrand and Marie-Joëlle Rochet for providing data and information on the northern Humboldt ecosystem and the Bay of Biscay, respectively, and for having participated so actively in the efforts of the IndiSeas WG. We also thank Francis Laloë and Luis Tito de Morais for their input at the beginning of the project, and Pierre Lopez for providing the world map (Figure 1). The clarity of the paper was improved following reviews from Keith Sainsbury and Jake Rice. MC was supported financially by post-doctoral fellowships from the Spanish Ministry of Science and Technology, and by a European Commission Marie Curie Post-doctoral Fellowship through the International Outgoing Fellowships (Call FP7-PEOPLE-2007-4-1-IOF) for the ECOFUN project. The contribution of HO was financed by a fundamental research grant from the Estonian Ministry of Education and Research (grant SF0180005s10). LJS was supported financially by the MEECE project and Astrid Jarre's South African Research Chair in Marine Ecology and Fisheries.

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Appendix

Data and expertise sources

A list of the IndiSeas data sources is provided in Table A1, and a list of the scientific experts that contributed knowledge and interpretation in Appendix Table A2.

Brief descriptions of the ecosystems compared in the first phase of IndiSeas

Adriatic Sea (north-central)

The north-central Adriatic Sea has the widest continental shelf in the Mediterranean Sea (Pinardi *et al.*, 2006). Through river run-off and oceanography, the region exhibits a decreasing trend in nutrient concentration and production from north to south and from west to east (Zavatarelli *et al.*, 1998). The area is important for the spawning of small pelagic fish (e.g. anchovy, *Engraulis encrasicolus*, and sardine, *Sardina pilchardus*) which constitute the bulk of purse-seine catches (Arneri, 1996; Agostini and Bakun, 2002). The demersal fisheries take juveniles of several target species, such as hake, *Merluccius merluccius*, red mullet, *Mullus barbatus*, and flatfish, and invertebrates such as the Norway lobster, *Nephrops norvegicus*, also yield an important proportion of the catch. Important changes in the small pelagic fish community have been recorded from the late 1970s, with a collapse of anchovy and a decrease in other small pelagic species in the area. Ecological perturbations in the ecosystem include changes in species composition attributable to exploitation, proliferation of species and invasions, change in species distribution as a result of environmental changes, harmful algal blooms, and eutrophication (Legovic and Justic, 1997; Galil, 2000; Santojanni *et al.*, 2006; Bombace and Grati, 2007).

Baltic Sea (central)

The Baltic Sea is one of the largest brackish water areas in the world and is connected to the North Sea via a single narrow and shallow strait. Its semi-enclosed nature creates a north–south salinity gradient and a permanent haline stratification. Once oligotrophic, the Baltic Sea has become eutrophic and contaminated during the second half of the 20th century, with cyanobacterial blooms as a common late-summer event (HELCOM, 2002, 2003, 2006). The oceanographic conditions in the Baltic are governed mainly by meteorological conditions influencing saline water inflow from the North Sea (Hänninen *et al.*, 2000). Strong inflow was frequent until the mid-1970s, but much rarer thereafter, which has caused stagnation periods in the deep basins (ICES, 2004, 2007a). Its main pelagic foodweb is relatively simple, consisting of a few dominant copepod species (*Pseudocalanus*, *Temora*), and two clupeids. There are three internationally assessed and managed commercial fish species in the area: herring, *Clupea harengus membras*, sprat, *Sprattus sprattus*, and cod *Gadus morhua callarias*, for which the population dynamics are significantly influenced by climate-driven changes in the environment (Köster *et al*., 2003, 2005; MacKenzie and Köster, 2004; Casini *et al*., 2006). Coupled with high fishing pressure during the past two decades, a shift from a demersal- to a pelagic-dominated system has taken place (Möllmann *et al*., 2006).

Barents Sea

The Barents Sea is a high-latitude shelf ecosystem, covered by ice in the northeast in winter. There is an inflow of warm Atlantic water and coastal water from the west, and fresh, cold Arctic water flows in from northeast. The Atlantic and Arctic water masses meet along the Polar Front, which is characterized by strong gradients in both temperature and salinity. There is great interannual variability in ocean climate related to the variable strength of the Atlantic water inflow and the exchange of cold Arctic water. Therefore, seasonal variation in hydrographic conditions is quite large. It is a spring-bloom system. The algal bloom along the Polar Front sustains a large productivity of zooplankton, which in turn supports large stocks of fish, seabirds, and marine mammals. The most abundant commercially exploited fish species are cod, haddock, *Melanogrammus aeglefinus*, and capelin, *Mallotus villosus*. There is also a large stock of immature herring, *C. harengus*, which migrates out of the system when mature. Capelin is a key species in the system, feeding on zooplankton and serving as a major transporter of biomass from the northern to the southern Barents Sea. The stock has, however, exhibited large fluctuations in abundance, including three collapses, during the past 25 years. The collapses were caused by overfishing, predation on maturing capelin by cod, predation on capelin larvae by juvenile herring of the strong 1983 year class (the first collapse was in 1985), and strong herring year classes preying on capelin larvae and cod preying on larger capelin (in 1993 and 2003).

Bay of Biscay

In the Bay of Biscay, the diversity of coastal habitat and the mixing of temperate with subtropical and boreal waters favour great species diversity. Living resources in the Bay of Biscay include more than 100 species of fish, cephalopods, and crustaceans, most of which are exploited by multispecies fleets based mainly in France, Portugal, and Spain. In coastal areas, demersal and benthic resources are exploited with a diversity of fishing gears. Offshore, trawling is important, but fixed gears are used increasingly (ICES, 2007a). The Bay of Biscay fish community has been affected by fishing for a long time. A number of top predator species was depleted in the early 20th century (Quéro and Cendrero, 1996). Trawls with small mesh size are used, catching large quantities of small fish (ICES, 1991a), of which many are discarded (Rochet *et al*., 2006, and unpublished onboard observer data, 2002–2006). Using survey-based indicators for the whole community and 51 target and non-target fish populations, large changes were detected between 1987 and 2002 which could not be ascribed to a reduced impact of fishing (Rochet *et al*., 2005), so the fish community remains heavily impacted by fishing and dominated by small species (SIH-C, 2007).

Bering Sea and Aleutian Islands

The Bering Sea and Aleutian Islands are considered to be a single ecosystem because many of the stocks that are commercially important in the two systems are assessed as a unit. The eastern Bering Sea covers an area of $\sim$500 000 km^2 (Aydin and Mueter, 2007; Aydin *et al*., 2007), and the Aleutian Islands extends from 170°W to 170°E out to the 500-m isobath, encompassing an area of $\sim$60 000 km^2. The total area of the combined systems is therefore $\sim$560 000 km^2 (Heymans, 2005). The Bering Sea is a dynamic ocean characterized by intense storms and substantial seasonal ice cover, the extent and nature of which affects all levels of the biological system. Large-scale changes in the system have been linked to the Pacific Decadal Oscillation (PDO; Hare and Mantua, 2000), and specifically for the Aleutian Islands, it seems that there is a negative correlation between the PDO and the general trends in the main species of the ecosystem (Heymans *et al*., 2007). Recently, the system seems to have changed from one dominated by cold-water, Arctic species to a temperate system in which a new set of species might dominate (BEST, 2004). The Bering Sea supports the United States' most productive and valuable fisheries, massive populations of marine birds and mammals, and subsistence activities of Native American communities (BEST, 2004). The system provides rich food resources for large populations of higher-level resident (walleye pollock, *Theragra chalcogramma*, flatfish, and shellfish) and transient (Pacific salmon, *Oncorhynchus* spp., seabirds, and marine mammals) taxa (Springer, 1992; Brodeur *et al*., 1999; Loughlin *et al*., 1999). The Aleutian Islands have been an important historical fishing ground for non-US vessels, and in the early 1960s, Japanese and Soviet fisheries expanded to the eastern Bering Sea and Aleutian Islands and started fishing for yellowfin sole, *Limanda ferruginea*, and Pacific ocean perch, *Sebastes alutus* (Forrester *et al*., 1978). After the decline in that fishery in 1972, the fishery turned to walleye pollock and Atka mackerel, *Pleurogrammus monopterygius*. Sablefish, *Anoplopoma fimbria*, Pacific cod, *Gadus macrocephalus*, arrowtooth flounder, *Atheresthes stomias*, Kamchatka flounder, *Atheresthes evermanni*, and Greenland halibut, *Reinhardtius hippoglossoides*, are also important in the trawl fisheries of the area (Forrester *et al*., 1978).

Benguela (southern)

The southern Benguela upwelling system is separated from the northern Benguela by a permanent upwelling centre near Lüderitz, Namibia. It extends from 29°S to 28°E to depths of 500 m, covering 220 000 km^2. A unique feature is that it also includes the Agulhas Bank region on South Africa's south coast, rendering the demersal and benthic parts of the foodweb more important than in most other upwelling systems. During the 1980s, anchovy, *E. encrasicolus*, was the dominant small pelagic fish, but in the 1990s, sardine, *Sardinops sagax*, round herring, *Etrumeus whiteheadi*, horse mackerel, *Trachurus t. capensis*, and Cape hake, *Merluccius capensis* and *M. paradoxus*, increased in abundance, the last three being mainly demersal. Anchovy and sardine both attained unusually high stock sizes in the early 2000s, although the situation was short-lived and sardine, in particular, has declined dramatically in recent years. The Benguela supports important pelagic and demersal fisheries; the purse-seine fishery targets anchovy and sardine, the midwater trawl fishery targets adult horse mackerel, and demersal trawlers target hake (commercially the most important, and also sustaining a longline fishery) and catch several other important bycatch species. It is believed that the demersal fishery is at maximum capacity and that most inshore linefish stocks are currently overexploited.

Canada west coast (Vancouver Island)

The west coast of Vancouver Island, Canada, includes the widest continental shelf along the west coast of North America south of the Aleutian Islands ($\sim$100 km). It is also at the northern extent of the California upwelling zone (Ware and McFarlane, 1989) and experiences seasonal (spring/summer) upwelling. The central and outer parts of the continental shelf undergo reversals in the main current direction (north-flowing in winter; south-flowing in summer), whereas there is a year-round northward flow at mid-depth (the California Undercurrent). The inner (near-coastal) part of the region consists of a persistent north-flowing current driven by freshwater run-off (Thomson *et al.*, 1989). The area is highly productive of plankton and fish (McFarlane *et al.*, 1997). Key resident fish species of commercial interest include Pacific herring (*Clupea pallasi*), Pacific cod, pandalid shrimps (e.g. *Pandalus jordani*), Pacific halibut (*Hippoglossus stenolepis*), and many species of rockfish (Scorpaenidae). The region also hosts a large biomass of migratory species, including Pacific hake (*Merluccius productus*), Pacific sardine (*S. sagax*), the abundance of which vary depending on warm or cool regimes, and various species of Pacific salmon, marine mammals, and seabirds. Scientific assessments suggest that most commercial species are fully exploited.

Catalan Sea (southern)

The southern Catalan Sea is located in the northwestern Mediterranean Sea, in the Catalano–Balearic basin. It has a wide continental shelf influenced by the Ebro River delta. Most of the data presented in our study refer to material from the continental shelf and upper slope, covering a total area of soft bottom sediments of 5000 km^2. The southern Catalan Sea ecosystem is oligotrophic, but enrichment by regional, environmental events is related to wind conditions, the temporal thermocline, the shelf-slope current, and river discharges (Estrada, 1996). The area is especially relevant to the reproduction and catches of small pelagic fish, mainly anchovy, *E. encrasicolus*, and sardine, *S. pilchardus* (Palomera *et al.*, 2007). Official landings increased from the beginning of the 19th century to the 1990s mainly through the expansion of the fishery and public incentives to the fishing sector. From the 1970s, however, there were marked fluctuations, and landings underwent a progressive decrease from 1994 to the present (Coll *et al.*, 2008a, b). Fishing activities target mainly small pelagic fish, juveniles of demersal species such as hake *M. merluccius*, and invertebrates. Ecological changes in the area include changes in species abundance and composition as a consequence of exploitation and changes in species distribution as a result of environmental events (Bas *et al.*, 2003; Lloret *et al.*, 2004; Sabatés *et al.*, 2006). Scientific assessments suggest that several demersal stocks are fully to over-exploited and that some pelagic stocks also show signs of overexploitation (Sardà, 1998; Palomera *et al.*, 2007).

Guinean EEZ

The EEZ of Guinea has an area of 42 917 km^2, with a large part shallow at 20–40 m (Domain and Bah, 1993). The Guinean coastline is 300 km long and receives freshwater from a dozen rivers, including a dense network of waterways that pours into the sea through mangroves. The marine ecosystem off Guinea is highly energetic, under the influence of mangroves and terrigenous input by the coastal rivers, but upwelling also plays a key role in the north (Baran, 1999; Pezennec, 1999). The Guinean marine fish community has been described by Domain (1980a, b), and the ecology by Domain *et al.* (1999), Jouffre and Domain (1999a, b), and Guénette and Diallo (2004). Guinean marine resources are exploited by an industrial fleet of foreign vessels operating under licence and by a national small-scale fleet including both traditional fishing canoes and modern motorized canoes (Domain *et al.*, 1999). Although the heavy exploitation is a relatively recent phenomenon compared with that of its neighbouring countries, Senegal and Mauritania, Guinean fisheries resources are currently regarded as heavily exploited, and in particular most of the coastal stocks are considered to be overfished (Lobry *et al.*, 2003; Sidibe, 2003; Chavance *et al.*, 2004).

Humboldt (northern, Peru)

The northern Humboldt Current System off Peru produces more fish per unit area than any other region of the world, despite representing $<0.1\%$ of the world's ocean surface; currently it produces $\sim$10% of the global wild fish catch (Chavez *et al.*, 2008). Like most other eastern boundary ecosystems, it is characterized by a general equatorward flow, intense coastal upwelling, and high levels of productivity, supporting the commercially important fish stocks (Wolff *et al.*, 2003). Upwelling of cool water brings nutrients to the surface, dramatically increasing the biological productivity in this low-latitude region of few storms (Chavez *et al.*, 2008). The region is notable for the *El Niño* phenomenon and climate variability in general. Through ocean/atmosphere coupling, the northern Humboldt Current is intimately linked to equatorial Pacific dynamics and is subject to large interannual to multidecadal fluctuations in climate, ecosystems, and fisheries (Chavez *et al.*, 2003, 2008; Bertrand *et al.*, 2008b; Fréon *et al.*, 2008; Gutiérrez *et al.*, 2009). Interannually and multidecadally, the system changes from one of high biological productivity and low diversity during cool periods to low productivity and high biodiversity during warm periods. Climate exercises bottom-up forcing at interannual (Barber and Chavez, 1983; Bertrand *et al.*, 2008a; Taylor *et al.*, 2008), multidecadal (Chavez *et al.*, 2003; Alheit and Ñiquen, 2004, Ayón *et al.*, 2008), and secular (Sifeddine *et al.*, 2008; Valdés *et al.*, 2008; Gutiérrez *et al.*, 2009) time-scales (Chavez *et al.*, 2008). Pelagic fish are targeted by purse-seiners and provide 95% of the Peruvian catches, the most important being anchoveta, *Engraulis ringens*, sardine, *S. sagax*, the transzonal jack mackerel, *Trachurus murphyi*, and the chub mackerel, *Scomber japonicus*. The anchoveta population is recognized as the largest neritic fish population and the largest single-fishery stock ever recorded. Peruvian hake (*Merluccius gayi peruanus*) is the main demersal species exploited.

Humboldt (southern, central Chile)

The southern Humboldt System extends approximately from 33 to 39°S and out to 30 nautical miles from shore (at the continental shelf break), with a total area of $\sim$50 000 km^2. As a geographical unit, it corresponds to the Mediterranean District and is ecologically independent from the Peruvian Province and the Austral District located north and south, respectively. The system is characterized by a narrow continental shelf ($<$30 nautical miles), seasonal upwelling (September–March), a shallow oxygen minimum zone (>0.5 ml O_2 l^{-1}), high primary productivity (19.9 g C m^{-2} d^{-1}), and globally significant landings ($>$4.5 million tonnes in 1995; Strub *et al.*, 1998; Daneri *et al.*, 2000; Escribano *et al.*, 2003). The ENSO cycle (interannual) and regime shifts (interdecadal) produce the main large-scale climate variability (Montecinos *et al.*, 2003; Alheit and Ñiquen, 2004). The plankton is dominated by large diatoms, copepods, and euphausiids. Macrocrustaceans such as squat lobsters (*Pleuroncodes monodon*, and *Cervimunida johni*) and shrimps (*Heterocarpus reedi*) are commercially important benthic components. Plankton-feeding pelagic fish such as anchovy

(*E. ringens*), Araucanian herring (*Strangomera bentincki*), and horse mackerel (*Trachurus symmetricus*) support major fisheries. Chilean hake (*M. gayi*) is the main demersal resource, and it feeds on euphausiids, crustaceans, small pelagic fish, and small hake (cannibalism; Meléndez, 1984; Arancibia, 1989; Arancibia *et al.*, 1998; Cubillos *et al.*, 2003). Top predators are still poorly studied. Industrial fisheries started in the 1940s and after peaking in the mid-1990s, landings declined, and most resources are now categorized as fully or overexploited. In addition, the TL of the landings has declined recently (Arancibia and Neira, 2005).

Irish Sea

The Irish Sea lies between England and Wales, and Ireland and Northern Ireland, and covers ∼58 000 km^2, ∼51–55°N, and 3–6.5°W. A north-to-south-running deep-water channel (the St Georges Channel) with a maximum depth of 150 m separates its relatively shallow western and east coastal regions. The main flow of water through the deep western Irish Sea flows south to north, whereas an anticlockwise gyre dominates circulation patterns in the eastern Irish Sea. A number of biological changes have been observed coincident with the North Atlantic climatic regime shift and heavy fishing in the Irish Sea since the 1970s. Both fish and zooplankton appeared to decline from high to low biomass during the period 1979–1983, indicated by a shift from a positive to a negative phase in the North Atlantic Oscillation Index. The trophic web of the Irish Sea has been described in an ecosystem model by Lees and Mackinson (2007), and it supports valuable pelagic trawl (herring), demersal trawl (targeting Norway lobster, *N. norvegicus*, cod, haddock, whiting, *Merlangius merlangus*, and plaice, *Pleuronectes platessa*, bycatch including anglerfish, *Lophius* spp., hake, and sole *Solea solea*), inshore trawl (shrimp and flatfish), inshore net fisheries (sea bass, *Dicentrarchus labrax*, cod, grey mullet, *Chelon labrosus*, sole, plaice), and fisheries for brown crabs and lobsters (Pawson *et al.*, 2002). Owing to the changing abundance and species composition of the stocks, developments in fishing technology and the constraints imposed by management measures, Irish Sea demersal fisheries have altered markedly over time. Major events since the 1990s include the decline in cod and whiting stocks, the growth of the haddock stock, and the introduction of recovery measures for cod from 2000 on.

Mauritanian EEZ

The Mauritanian ecosystem (Atlantic Ocean) is another of the most productive upwelling systems in the world, extending from 16°04′ to 20°36′N, along 720 km of coastline with an EEZ of 234 000 km^2. The area has 31 700 km^2 of continental shelf (24 650 km^2 excluding the marine protected part of the Arguin Bank National Park). The ecosystem is characterized by permanent upwelling in the north and seasonal upwelling in the south, from January to April. The influences of both the Canarian and the Guinean Currents, respectively, southwards and northwards, make this an area of transition from temperate to tropical waters that supports high productivity and hence a great abundance of shelf marine resources as well as temperate and tropical species, depending on season. This has stimulated a rapid growth of fisheries since 1982. During the first 15 years of exploitation, the number of fishing vessels rose from 150 to 300–350, and dugouts increased in number from 580 to 3500 (IMROP, 2007). This led to an increase in industrial and small-scale fisheries effort and a concomitant decline in the abundance of pelagic resources such as the round sardinella, *Sardinella aurita*, and most demersal species, mainly cephalopods and some sparids. Demersal biomass dropped from an estimated 600 000 t in 1982 to 150 000 t in 2006, a 75% reduction in biomass and a decrease in mean TL (Gascuel *et al.*, 2007). The demersal resources are considered to be overexploited.

North Sea

The North Sea covers some 570 000 km^2 (Jones, 1982) of the European continental shelf with an average depth of ∼90 m, the deepest part in the Norwegian Trench, which is ∼400 m deep. It is bounded by the coasts of Norway, Denmark, Germany, the Netherlands, Belgium, France, and the UK and is recognized as an LME (McGlade, 2002). The continental coastal zone (mean depth 15 m) covers an area of ∼60 000 km^2 and is under a strong influence from terrigenous inputs. The North Sea lies in ICES Divisions IVa–c and is approximately bounded by the area 4°W–8°E and 51–62°N. The northern part is comparatively deep and subject to strong oceanic influence, characterized by seasonal stratification of the water column. The southern North Sea is shallower (20–50 m) and remains mixed for most of the year, except where there are significant freshwater inputs such as from the River Thames. The seafloor consists of mostly mixed sediments. Diversity is lower in the shallow southern North Sea and eastern Channel (Rogers *et al.*, 1998). Inshore, where there is more variation in sediment type and a more spatial patchiness, the species diversity is generally higher (Greenstreet and Hall, 1996). In all, 224 fish species have been recorded (Knijn *et al.*, 1993). The main fisheries can be split into demersal, pelagic, and industrial. Demersal fisheries target roundfish species such as cod, haddock, whiting, and saithe, *Pollachius virens*, in addition to flatfish species such as plaice and sole. Pelagic fisheries target herring and mackerel, *Scomber scombrus*, and the industrial fisheries target sandeel, *Ammodytes* spp., Norway pout, *Trisopterus esmarkii*, and sprat, *S. sprattus*, which are used to produce fishmeal and oil. There are also important crustacean fisheries for Norway lobster, shrimp, *Pandalus borealis*, and brown shrimp, *Crangon crangon*. The North Sea supplies approximately 2 million tonnes of fish each year from its industrial, pelagic, and demersal fisheries. Responsibilities for fisheries management in the North Sea lie both with neighbouring countries through economic exclusion zones and with the European Commission, which sets total allowable catches (TACs) for its countries under the principles of the Common Fisheries Policy. Scientific advice on the state of the stocks and proposed TACs is provided by the International Council for Exploration of the Seas (ICES).

Portuguese EEZ

The continental Portuguese ecosystem is situated in the northern region of the Canary Current upwelling. Portuguese continental waters (excluding Madeira and the Azores) stretch from 36.5 to 41.5°N, the country's EEZ has an area of ∼327 700 km^2, of which ∼23 700 km^2 is continental shelf. During spring and summer, northerly winds along the coast dominate, causing coastal upwelling (Wooster *et al.*, 1976; Fraga, 1981; Fiúza *et al.*, 1982). In autumn and winter, the surface circulation is predominantly northwards, partially driven by southerly winds and meridional alongshore density gradients (Peliz *et al.*, 2003, 2005), and transporting warmer (subtropical) water higher in salinity and poor in nutrients over the shelf break (Frouin *et al.*, 1990; Haynes and Barton, 1990; Ruiz-Villarreal *et al.*, 2006). There is evidence of northward and southward shifts in fish distribution associated with climate drivers (Stratoudakis *et al*., 2002). The area is important for small pelagic fish, mainly sardine, *S. pilchardus*, which accounts for between 50 and 80% of the catches; together with horse mackerel, *Trachurus trachurus*, and chub mackerel, *Scomber colias*, pelagic fish constitute ~80–90% of the total landings. Bottom trawlers target demersal species such as hake and invertebrates, mainly *N. norvegicus* and *Octopus vulgaris*. The distribution of the migratory snipefish, *Macrorhamphosus* spp., a non-commercial species associated with saline and warm water, extended north during the late 1990s. Long-term productivity cycles, based on historical catch time-series analysis, are associated with the North Atlantic Oscillation (NAO), supporting hake, sardine, horse mackerel, chub mackerel, anchovy, and blue whiting, *Micromesistius poutassou* (Borges *et al*., 2003). The structure of the fish community has been stable over time in terms of depth and latitude (Gomes *et al*., 2001; Sousa *et al*., 2005), despite varying species abundance and trends (Bianchi *et al*., 2000). Bottom-trawl effort and associated landings after the early 1950s, peaked in the mid-1970s, and since then has been decreasing. Single-species stock assessments show that hake and Norway lobster are overexploited, and recovery plans for both are in place (ICES, 2007b).

Morocco (Sahara coastal)

The Moroccan study area extends from Cape Bojador to Cape Blanc (21–26.25°N, 56 700 km^2) and represents part of the Canary Current upwelling, with the latter intense and permanent throughout the year. One of the main oceanographic features of the area is the boundary near Capes Barbas (~23°N) and Blanc (~21°N) between cold North Atlantic Central Water, transported south by the Canary Current, and warmer South Atlantic Central Water, transported north by the North Equatorial Counter Current. The marine community reflects these physical transitional characteristics, with subtropical species (>70%) coexisting with tropical and temperate ones. Small pelagic species, dominated by European sardine, form the bulk of the biomass in this upwelling area, Cephalopods, sparids, and sciaenids dominate the demersal community. The area has suffered uncontrolled international exploitation for decades, especially before the 1980s. Heavy exploitation of many long-lived piscivores (Gulland and Garcia, 1984; Caddy and Rodhouse, 1998) and discarding practices (Balguerías *et al*., 2000) seemingly modified the foodweb structure, resulting in a short-lived cephalopod outburst after the late 1960s. Small pelagic fish and cephalopods currently support most of the fisheries of the area.

Scotian Shelf (eastern), Canada

The eastern Scotian Shelf is a broad (~200 km, 90 m average depth) temperate continental shelf consisting of shallow offshore banks and inner basins. It extends from the Laurentian Channel in the northeast to a line from Halifax, NS, south to the shelf break in the southwest, an area of ~100 000 km^2. Its physical environment is governed by its location near the meeting place of major currents of the Northwest Atlantic, and its complex topography (Zwanenburg *et al*., 2006). Water and organisms are transported from the northeast towards the southwest, which in the mid-1980s precipitated a major cooling event of bottom waters. It is a productive ecosystem that supports vibrant groundfish fisheries. However, in the early 1990s, the cod stock collapsed and other groundfish species experienced serious declines. Overall, there has been a community-level reduction in body size, biomass, and physiological condition of resident demersal fish species (Choi *et al*., 2004). The ecosystem has switched from being dominated by demersal species to one dominated by forage species (Bundy, 2005), although the abundance of one top predator, the grey seal, *Halichoerus grypus*, has increased five-fold since the late 1980s. Since the groundfish fisheries moratorium was instituted in 1993, cod have not recovered, and the fishing industry has focused on lucrative invertebrate species including lobster, *Homarus americanus*, shrimp, *P. borealis*, and snow crab, *Chionoecetes opilio*. These fisheries bring in more revenue than groundfish fisheries, but the price is an ecosystem that has undergone a fisheries-induced regime shift with an accompanying trophic cascade (DFO, 2003; Bundy, 2005; Bundy and Fanning, 2005; Frank *et al*., 2005).

Senegalese EEZ

The EEZ of Senegal, located between a north–south orientated coastline some 700 km long and the 200-mile offshore limit, spans an area of ~158 900 km^2. It covers a continental shelf considered to be highly productive (Domain, 1980a, b; Cury and Roy, 1991; Caverivière *et al*., 2002) because of the influence of complex hydrology, involving seasonal coastal upwelling combined with two major oceanic currents (Canary and Guinea). Located at the transition between temperate and tropical fauna, Senegalese waters are populated by a wide range of species. Most of them, fish but also invertebrates (specially cephalopods, crustaceans, gastropods, and bivalves), have been subjected to heavy fishing pressure for decades from both industrial and artisanal fleets (Chavance *et al*., 2004; Jouffre *et al*., 2004). Today, the artisanal sector is particularly active, with a fleet exceeding 10 000 canoes (CRODT, 2006), by far the largest in the region. With an annual production of some 400 000 t, the fishing sector plays a strategic role in the Senegalese economy. Scientific assessments suggest that most demersal stocks are fully to overexploited, and several of the main pelagic stocks also show signs of overexploitation (Chavance *et al*., 2004; CRODT, 2005).

United States (northeast continental shelf)

The northeast US continental shelf LME includes the Gulf of Maine, Georges Bank, southern New England, and a mid-Atlantic region (Sherman, 1991), covering an area of ~293 000 km^2 from Cape Hatteras to the Nova Scotian waters of the Northwest Atlantic. It is a highly productive ecosystem (O'Reilly and Zetlin, 1998) that has supported significant commercial fisheries for centuries (Sissenwine *et al*., 1984). In general terms, its circulation is mainly from the northeast to the southwest, with an anticlockwise flow around the Gulf of Maine and a clockwise flow on Georges Bank, and some longshore flow from the south up onto the shelf. The recent history of the fish community has exhibited the classic cycles of excess effort, stock decline, and iterations thereof, until the point of sequential stock depletion (Serchuk *et al*., 1994; Murawski *et al*., 1997; Fogarty and Murawski, 1998). Currently, most demersal stocks are at moderate to low levels, elasmobranch stocks are declining, and small pelagic stocks are at record highs (Serchuk *et al*., 1994; Fogarty and Murawski, 1998; Link and Brodziak, 2002; Overholtz, 2002). The foodweb has a disproportionately large number of species interactions (Link, 2002). Along with the various fisheries and their effects, there were notable changes to protected species, with many more now in a critical condition than there were 50 years ago (Waring *et al*., 2004). Additionally, there have been shifts in non-targeted fauna (Link and Brodziak, 2002).

Appendix Table A1. Source of the data for calculating indicators.

	Survey and biomass data		Catch data		Source of species parameters		
Ecosystem	**Source**	**Time-series**	**Source**	**Time-series**	**Maximum lifespan**	**TL**	**Stock status**
Adriatic Sea (north-central)	Bottom trawling, acoustic surveys by ISMAR–CNR	1976–2006	Italian Government (ISTAT, IREPA); ISMAR–CNR	1975–2006	Fishbase[a]	Ecopath models (Coll *et al.*, 2007, 2009)	FAO (2005)[b]; ISMAR–CNR
Baltic Sea (central)	ICES (2006a)	1974–2005	ICES (2006a)	1974–2005	ICES (2006a)	Expert opinion	ICES[c]
Barents Sea	ICES (2006b, c, 2007c); Jakobsen *et al.* (1997)	1984–2006	FAO statistics	1984–2006	Fishbase[a]; expert opinion	Ecopath model Skaret and Pitcher (in press)	FAO (2005)[b]; ICES[c]; expert opinion
Bay of Biscay	Survey EVHOE on board RV "Thalassa" conducted annually in autumn in the Bay of Biscay; ICES (1991b)	1994–2005	ICES[c]	1993–2005	ICES[c]	J. Lobry, pers. comm.	ICES[c]
Benguela (southern)	Small pelagic fish: November spawner-biomass surveys (MCM–DEAT); demersal species: swept-area demersal surveys (January for west coast, May for south coast) (MCM–DEAT); Cunningham and Butterworth (2004a, b); Shannon *et al.* (2008)	1986–2006	MCM–DEAT unpublished data; FAO statistics	1980–2006	Fishbase[a] and/or estimated from von Bertalanffy parameters	Ecopath models (Shannon *et al.*, 2003, 2004, 2008)	FAO (2005)[b]; T. Fairweather, pers. comm.; L. Hutchings, pers. comm.; Southern African Sustainable Seafood Initiative[d]
Bering Sea, Aleutian Islands	NOAA (2008)	1977–2006	NOAA (2008)	1977–2006	Alaska Fisheries Science Center[e]	Ecopath model (Heymans *et al.*, 2005)	FAO (2005)[b]
Canada, west coast	Bottom-trawl surveys using small-mesh gear conducted annually by Fisheries and Oceans Canada (Pacific Biological Station, Nanaimo)	1980–2007	Fisheries and Oceans Canada sources	1980–2005	Fishbase[a]	Fishbase[a]	Fisheries and Oceans Canada stock assessments[f]; local expertise
Catalan Sea (southern)	Bottom trawling, acoustic survey by ICM–CSIC and IEO	1978–2003	Catalan Government[g] ICM–CSIC	1976–2006	Fishbase[a]	Ecopath models (Coll *et al.*, 2006, 2008a, b)	FAO (2005)[b]; ICM–CSIC (I. Palomera, G. Merino, pers. comm.); Greenpeace (A.R. Martín, pers. comm.); WWF (S. Tudela, pers. comm.)
Guinean EEZ	RV "André Nizery" (CNSHB); RV "Général Lassana Conté " (CNSHB)	1985–2000 to 2001–2006	CNSHB database	1985–2006	Fishbase[a]; local expertise; grey literature	Fishbase[a]; local expertise; grey literature	FAO (2005)[b]; local expertise; grey literature
Humboldt (northern)	Small pelagics, acoustic surveys (IMARPE); demersal-species, swept-area surveys (IMARPE); VPA stock assessments (IMARPE)	1983–2006	IMARPE statistics	1983–2006	Fishbase[a]; Jordán (1976); Mendo (1984); Serra and Tsukayama (1988); Argüelles *et al.* (2001); Bouchon (2007); Fernandez, pers. comm.	Fréon *et al.* (2009); A. Bertrand and J. Tam, pers. comm.	FAO (2005)[b]; M. Ñiquen, pers. comm.
Humboldt (southern)	Technical reports published by the Chilean Fishery Research Fund (Fondo de Investigación Pesquera)[h]	1993–2005	Landings statistics yearbooks published by the Chilean National Fisheries Service (Servicio Nacional de Pesca)[i]	1993–2005	Technical description of target species published by the Chilean Undersecretary of fisheries[j]	Ecopath models (Neira and Arancibia, 2004; Neira *et al.*, 2004); Fishbase[a]	FAO (2005)[b]

Irish Sea	ICES Division VIIa (bass, cod, plaice, whiting, sole, herring, haddock from ICES stock assessments, all other estimates from Cefas Irish Sea beam-trawl survey)	1980–2005	ICES Statlan database extracted for ICES Division VIIa 1973–2005	1973–2003	Fishbase[a]	Ecopath model (Lees and Mackinson, 2007)	FAO (2005)[b]; ICES[c]
Mauritanian EEZ	Experimental demersal bottom trawls (IMROP RV)	1982–2007	IMROP database (IMROP, 2007)	1990–2005	Fishbase[a]	Fishbase[a]; Laurans *et al.* (2004); Sidi and Guénette (2004)	FAO (2006); IMROP (2007); Gascuel *et al.* (2007)
Morocco (Sahara coastal)	Small pelagics: hydroacoustic surveys on RV "Dr Fridtjof Nansen". Demersal species: bottom trawl RV "Charif Al Idrissi"	1998–2005	FAO database	1993–2005	Fishbase[a]; local expertise	Fishbase[a]	CECAF; FAO (2005)[b]
North Sea	North Sea International Bottom-Trawl Survey data (IBTS) from ICES (biomass); North Sea English Groundfish Survey (EGFS) (length data)	1983–2006	ICES WG reports; ICES Statlan database	1963–2003	Fishbase[a]	Ecopath model (Mackinson and Daskalov, 2007)	FAO (2005)[b]; ICES[c]
Portuguese EEZ	Survey data 1982–2006; DGPA statistics for landings data	1981–2006	ICES reports; Portuguese Fisheries Service (DGPA)	1981–2006	Fishbase[a]; survey data	Fishbase[a]	ICES[c]
Scotian Shelf (eastern)	DFO multispecies research surveys (bottom trawl); 1970–1981 RV "A.T.Cameron"; 1982 RV "Lady Hammond"; 1983–present RV "Alfred Needler"; All data housed in DFO Maritimes Virtual Data Centre	1970–2006	DFO catch statistics	1960–2006	Fishbase[a]; Scott and Scott (1988)	Bundy (2004)	FAO (2005)[b]; DFO Canadian Stock Assessment Secretariat reports[k]
Senegalese EEZ	RV "Louis Sauger" (CRODT); RV "Itaf Deme" (CRODT)	1981–2000; 2001–2005	CRODT database	1981–2005	Fishbase[a]; local expertise; grey literature	Fishbase[a]; local expertise; grey literature	Fishbase[a]; local expertise; grey literature
US coast (northeast)	NEFSC bottom-trawl survey database (Azarovitz, 1981; NEFC, 1988)	1963–2007	NMFS Fisheries Statistics Database	1964–2005	NEFSC survey database and ageing programme (Penttila and Dery, 1988); Fishbase[a]	NEFSC Food Habits Database (Link *et al.*, 2000); Fishbase[a]; A. Bundy, pers. comm.	NEFSC stock assessments (SOS[l])

[a]http://www.fishbase.org.
[b]http://www.fao.org/docrep/009/y5852e/Y5852E10.htm#tbl.
[c]http://www.ices.dk.
[d]http://www.wwf.org.za/sassi.
[e]http://www.afsc.noaa.gov/refm/age/Stats/Max_age.htm and http://access.afsc.noaa.gov/reem/LHWeb/Index.cfm.
[f]http://www.csas.dfo-mpo.gc.ca.
[g]http://www.gentcat.cat/darp/.
[h]http://www.fip.cl.
[i]http://www.sernapesca.cl.
[j]http://www.subpesca.cl.
[k]http://www.meds-sdmm.dfo-mpo.gc.ca/csas/applications/Publications/publicationIndex_e.asp.
[l]http://www.nefsc.noaa.gov/sos/.

Appendix Table A2. Local scientific experts who participated in the IndiSeas project.

Ecosystem	Expert	Affiliation	e-mail address
Adriatic Sea (north-central)	Marta Coll	Institute of Marine Science (ICM – CSIC), Barcelona, Spain	mcoll@icm.csic.es
Baltic Sea (central)	Henn Ojaveer	Estonian Marine Institute, University of Tartu, Estonia	henn.ojaveer@ut.ee
Baltic Sea (central)	Christian Möllmann	Institute for Hydrobiology and Fisheries Science, University of Hamburg, Germany	christian.moellmann@uni-hamburg.de
Barents Sea	Edda Johannesen	Institute of Marine Research, Bergen, Norway	eddaj@imr.no
Bay of Biscay	Marie-Joëlle Rochet	Ifremer, Nantes, France	marie.joelle.rochet@ifremer.fr
Benguela (southern)	Lynne Shannon	Marine Research Institute and Zoology Department, University of Cape Town, South Africa	lynne.shannon@uct.ac.za
Benguela (southern)	Dawit Yemane	Marine and Coastal Management, Department of Environmental Affairs and Tourism, Cape Town, South Africa	dawityemane@gmail.com
Bering Sea, Aleutian Islands	Sheila (J. J.) Heymans	Scottish Association for Marine Science, Dunstaffnage Marine Laboratory, Oban, UK	sheila.heymans@sams.ac.uk
Bering Sea, Aleutian Islands	Kerim Aydin	Alaska Fisheries Science Center, Seattle, WA, USA	kerim.aydin@noaa.gov
Canada coast (West)	Ian Perry	Fisheries and Oceans Canada, Pacific Biological Station, Nanaimo, BC, Canada	ian.perry@dfo-mpo.gc.ca
Catalan Sea (southern)	Marta Coll	Institute of Marine Science (ICM – CSIC), Barcelona, Spain	mcoll@icm.csic.es
Guinean EEZ	Ibrahima Diallo	Centre National des Sciences Halieutiques de Boussoura (CNSHB), Conakry, Guinea	idiallo@cnshb.org
Guinean EEZ	Didier Jouffre	Institut de Recherche pour le Développement (IRD), UMR 5119, University of Montpellier II, France	didier.jouffre@ird.fr
Humboldt (northern)	Erich Diaz	Instituto del Mar del Perú, Lima, Perú	ediaz@imarpe.gob.pe
Humboldt (southern)	Sergio Neira	Centro de Investigación en Ecosistemas de la Patagonia, Coyhaique, and Departamento de Oceanografía, Universidad de Concepción, Concepción, Chile	se_neira@hotmail.com
Irish Sea	Julia Blanchard	Centre for Environment, Fisheries and Aquaculture Science, Lowestoft, UK	julia.blanchard@cefas.co.uk
Irish Sea	Steve Mackinson	Centre for Environment Fisheries and Aquaculture Science, Lowestoft, UK	steven.mackinson@cefas.co.uk
Irish Sea	John Cotter	Formerly Centre for Environment Fisheries and Aquaculture Science, Lowestoft, UK	john.cotter@phonecoop.coop
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doi:10.1093/icesjms/fsp294