Ranking the ecological relative status of exploited marine ecosystems

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A set of simple, data-based ecological indicators was used to rank exploited ecosystems regarding fishing impacts with respect to their status, trends, and ecosystem EAF attributes. Expected theoretical changes in indicators with respect to increasing fishing impacts were considered, and ecosystems were compared by examining the mean values of indicators in the most recent three years for which data were available and over time (1980–2005 and 1996–2005). Systems were classified into nine potential categories according to whether they were most, moderately, or least impacted, and whether they were becoming more or less impacted, or remaining stationary. The responses of ecological indicators to additional environmental and socio-economic explanatory factors were tested. Ecosystems ranked using short- and long-term trends and states differed because of differences in trends, underscoring the importance of analysing both states and trends in ecosystem analyses. The number of ecosystems classified as unclear or intermediately impacted has increased recently, the proportion of ecosystems classified as less strongly impacted has been maintained, but more now fall within the category more strongly impacted in terms of long-term trends and states. Ecosystem type, fisheries enforcement, primary production, sea temperature, and fishing type were important variables explaining the ecological indicators. The results reflect different changes and processes in the ecosystems, demonstrating that information on ecological, environmental, and fishery histories is crucial to interpreting indicators correctly, while disentangling the effects of fishing and of the environment.

Keywords: comparative approach, ecosystem approach to fisheries, ecosystem indicators, fishing impacts, multivariate analysis, ranking techniques.

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Introduction

Historically, fishing has been one of the most important human uses of the ocean. Yet it has led to significant declines of predatory organisms (Pauly *et al*., 1998; Cury and Cayré, 2001; Christensen *et al*., 2003), and because of its direct and indirect impacts, there have been substantial modifications of marine ecosystems worldwide (Walsh, 1981; Jackson *et al*., 2001; Lotze *et al*., 2006). Although the current primary objective of fisheries management is to ensure sustainable levels of commercial stocks, the incorporation of broader ecosystem considerations into managing fisheries is now crucial (Murawski, 2000; Link, 2002a, 2005; FAO, 2003; Garcia *et al*., 2003; Pikitch *et al*., 2004; Walters *et al*., 2005). The need for an ecosystem-based fishery management approach is now recognized by government entities, e.g. the European Community 2371/2002 (EC, 2002), Canada's Oceans Act (DOJ, 1996), and the US Commission on Ocean Policy (2004).

To progress towards a real implementation of the ecosystem approach to fisheries (EAF), carefully selected, simple, widely agreed ecological, economic, and social indicators are needed to quantify the ecosystem impacts of fishing and eventually to translate them into management actions. Ecological indicators are needed to quantify the impacts of fishing relative to other ecosystems and to use when providing scientific advice on management actions in the light of objective functions that society wishes to maximize [e.g. conservation of functional biodiversity (CB), sustainable exploitation]. Progress has included identifying, testing, and verifying which indicators are useful in assessing ecosystem effects of fishing, defining reference levels and reference directions for selected indicators, developing and testing evaluation frameworks, and proposing alternative ecosystem-based ecological indicators to incorporate into the fisheries management advice criteria (Pauly *et al.*, 1998; Link, 2002b, 2005; Cury *et al.*, 2005; Fréon *et al.*, 2005; Fulton *et al.*, 2005; Jennings and Dulvy, 2005; Rice and Rochet, 2005; Shin *et al.*, 2005; Trenkel *et al.*, 2007; Libralato *et al.*, 2008).

In 2005, a follow-up to the SCOR/IOC Working Group on Quantitative Ecosystem Indicators was initiated in the form of the EUR-OCEANS IndiSeas Working Group to undertake a comparative study on EAF ecological indicators (Shin and Shannon, 2010). A suite of community- to ecosystem-level indicators was agreed upon with respect to several criteria (ecological meaning, sensitivity to fishing, data availability, ecological objectives, and public awareness), selected to represent a minimum list of indicators that are easy to calculate. Indicators were quantified for several exploited ecosystems, and comparative results have provided insights on the relative current states and recent trends (Blanchard *et al.*, 2010; Shin *et al.*, 2010b) of several marine ecosystems located worldwide. It is important to note that this exercise had the relevant scope to evaluate the relative status of different exploited ecosystems using simple and available indicators, so as to include also ecosystems that are normally not included in studies using more complex indicators that are only applicable to data-rich situations.

Here, we provide an example of how, in the context of the EAF, this suite of ecological indicators can be used to rank and assess in a comparative context the relative ecological status of several marine ecosystems subjected to fishing. We build on the work on relative states and recent trends to explore several options to ranking and scoring the ecological status of these ecosystems using multiple indicators. We then evaluate this ranking, accounting for a set of ecosystem EAF attributes, and considering both longer (1980–2005) and shorter (1996–2005) time-series. Then, the comparative ranking is used to group ecosystems into lightly impacted, ecosystems that are moderately or lightly exploited and notably improving, or heavily impacted, such as moderately or highly impacted ecosystems that are deteriorating or not improving over time. It is important to understand that these analyses were undertaken in the absence of clearly defined reference points or baselines. Being able to compare our indicators with reference points and baseline levels would be the ideal situation. However, to meet our objective of comparing across a broad spectrum of exploited marine ecosystems, many of which are data-poor, alternative approaches to the problem had to be used. Much research is still required in this field, notably using ecosystem models, but the scientific community is still far from being able to propose the reference points, and models are simply unavailable for several ecosystems. Shin *et al.* (2010b) attempted to define reference points by surveying experts in some of the ecosystems to help define ranges of potential reference points for the indicator suite.

Finally, ranking of the ecosystems is compared on a multivariate basis, including potential environmental and socio-economic explanatory factors (or abiotic indicators). General patterns in ecosystem status at a broad scale are hence analysed and interpreted taking into account ecological, environmental, and socio-economic features, and the knowledge of experts on each ecosystem. The ranking results are evaluated by comparing them with partial information available from specific ecosystems and with previous comparative approaches using models and other indicators (Coll *et al.*, 2006a, 2008c; Libralato *et al.*, 2008; Shannon *et al.*, 2009a, b). Translating the values or trends of a set of ecological indicators into management decisions is an important yet difficult step towards making EAF a reality. What we hope to accomplish through the ranking process here is to integrate the information of various simple indicators and facilitate an EAF by providing a source of information to initiate serious efforts to take ecosystem considerations into account in managing the fisheries.

Methods

Case studies

In all, 19 exploited ecosystems were included in this analysis (Table 1). They corresponded to upwelling, high-latitude, temperate, and tropical marine ecosystems, and they covered a range of low- to highly productive areas, located in the Atlantic, Pacific, and Indian Oceans, as well as in the Mediterranean Sea. A description of each ecosystem is provided in Shin *et al.* (2010a).

Selection of ecosystem indicators

Eight indicators were used to rank the exploited ecosystems (Table 2; see Appendix for precise formulation). The mean length (labelled "fish size") and mean lifespan ("lifespan") of surveyed species, the proportion of predatory fish ("percentage of predators"), and the trophic level of landed catches ("trophic level") were used in the analysis of states and trends. The total biomass of surveyed species ("biomass/landings") and landings ("inverse fishing pressure") was used for trend analyses only. The coefficient of variation (CV) of total biomass (expressed as $1/CV$ of total biomass, "biomass stability") and the proportion of under- to moderately exploited stocks ("percentage of healthy stocks") were included only in the analysis of states. All ecological indicators were formulated to decrease with a higher impact of fishing, so the lower the value of the indicator or any decrease in the indicator over time, the greater the impact on the ecosystem as a result of fishing. The motivation on how these indicators were selected to be useful as communication tools and a detailed description of how to quantify and analyse them is presented in Shin and Shannon (2010) and Shin *et al.* (2010a). A description of the origin of the data and normalized datasets are presented in Shin *et al.* (2010a) and Blanchard *et al.* (2010).

States, trends, and ecosystem attributes

The mean values of the ecological indicators in the recent three years for which data were available (for most systems 2003–2005) were used to provide information on the current ecosystem state and to rank the ecosystems accordingly. Trends in indicators were examined over the 10 years 1996–2005 and also for a longer time-series (1980–2005), or for the years within this period for which data were available (Link *et al.*, 2010). Time-series of data were standardized $[(I_y - \bar{I})/\text{s.d.}(I)]$, where I_y is the value of indicator I in year y, $\bar{I}$ the mean of indicator I for the time-series included in the analysis, and s.d.(I) the standard deviation of indicator I for the time-series included in the analysis to allow comparison of trends. A simple linear model was used to describe trends by testing for autocorrelation, following Coll *et al.* (2008a) and Blanchard *et al.* (2010). We assessed the significance

Table 1. Ecosystems considered in the comparative approach and environmental and socio-economic information.

Ecosystem	Geographic area	Large marine ecosystem	Ecosystem type	PP[a]	SST[b]	Fisheries[c]	Enforcement[d]	HDI[e]
North-central Adriatic Sea	C Mediterranean	Mediterranean	Temperate	406	17.9	A&I	L–M	0.903
Central Baltic Sea	NE Atlantic	Baltic Sea	Temperate/ brackish	1 849	8.0	I	M–H	0.903
Barents Sea	NE Atlantic	Barents Sea	High latitude	437	1.7	I	M–H	0.920
Bay of Biscay	NE Atlantic	Iberian Coastal	Temperate	665	15.0	A&I	L–M	0.951
Southern Benguela	SE Atlantic	Benguela Current	Upwelling	1 340	17.9	A&I	L–M	0.674
Bering Sea	NE Pacific	East Bering Sea	High latitude	766	4.9	I	H	0.877
West coast Canada	E Central Pacific	Gulf of Alaska	Temperate	771	10.5	I	M–H	0.961
Southern Catalan Sea	NW Mediterranean	Mediterranean	Temperate	406	19.0	A&I	L–M	0.949
Southern Humboldt (Chile)	SE Pacific	Humboldt Current	Upwelling	826	13.2	A&I	M	0.867
Guinean EEZ	E Central Atlantic	Guinea Current	Tropical	908	27.7	A&I	L	0.456
Irish Sea	NE Atlantic	Celtic-Biscay Shelf	Temperate	884	11.7	I	H	0.953
Mauritania	E Central Atlantic	Canary Current	Tropical	1 198	23.8	A&I	L–M	0.550
Morocco (Sahara coastal)	E Central Atlantic	Canary Current	Tropical	1 198	22.0	A&I	L	0.646
North Sea	NE Atlantic	North Sea	Temperate	1 046	10.5	I	H	0.954
Northern Humboldt (Peru)	SE Pacific	Humboldt Current	Upwelling	826	18.7	A&I	L–M	0.773
Portuguese EEZ	NE Atlantic	Iberian Coastal	Temperate	665	17.2	A&I	M	0.897
Eastern Scotian shelf	NW Atlantic	Scotian Shelf	Temperate	1 269	8.3	I	M	0.961
Senegalese EEZ	E Central Atlantic	Canary Current	Tropical	1 198	25.5	A&I	L	0.499
Northeastern US (NEUS)	NW Atlantic	NEUS continental shelf	Temperate	1 451	15.6	I	H	0.951

[a]Annual mean value of 1998 from large marine ecosystem in mg C m^{-2} d^{-1}.
[b]Mean annual SST (°C) 2003–2005.
[c]Fisheries type, mainly industrial (I), both artisanal and industrial fisheries (A&I).
[d]Enforcement of fishery management, i.e. high (H), moderate (M), or low (L).
[e]HDI from 2006.

Table 2. Ecosystem indicators used in the IndiSeas project and the corresponding ecosystem EAF attributes.

Indicator	Label	Used for state (S) or trend (T)	Ecological goal
Proportion of underexploited or moderately exploited stocks	% healthy stocks	S	CB
Proportion of predatory fish	% predators	S, T	CB
1/coefficient of variation in total biomass	Biomass stability	S	SR
Mean lifespan	Lifespan	S, T	SR
Mean length of fish in the community	Fish size	S, T	EF
Trophic level of landed catch	Trophic level	S, T	EF
Total biomass of survey species	Biomass	T	RP
Biomass/landings	Inverse fishing pressure	T	RP

All indicators are expressed so that a decline in an indicator is expected when an ecosystem becomes more impacted by fishing (see Appendix for the formulation of indicators). CB, conservation of functional biodiversity; SR, ecosystem stability and resistance to perturbations; EF, ecosystem structure and functioning; RP, resource potential.

of the estimated trend (p-value) and the trend (positive or negative slope) and the magnitude of the slope to rank our ecosystems (reported in full in Blanchard *et al.*, 2010).

Four ecological goals were identified and each indicator was linked to one of them: (i) CB, (ii) ecosystem stability and resistance to perturbation (SR), (iii) ecosystem structure and functioning (EF), and (iv) resource potential (RP)—see Table 2 and Shin *et al.* (2010a). These ecosystem attributes were also used to perform the ranking, giving similar weight to each of the four attributes. Although there was some redundancy between indicators within most ecosystems, there was not a common pattern in redundancy between indicators across ecosystems (Blanchard *et al.*, 2010), so different weightings were not assigned to indicators to correct for redundancy and correlation.

Ranking exploited ecosystems

Ranking of the exploited ecosystems was performed following similar methodology to that used in a comparative analysis of fishing impacts, which was undertaken previously using synthetic, model-derived ecosystem indicators and ranking ecosystems in terms of their exploitation level (Shannon *et al.*, 2009a). Along similar lines, the ranking was performed by cross-comparing results in both recent years (ecosystem states), as well as over time (indicator trends), and accounting for the ecosystem attributes mentioned above.

Because indicators were expressed so that a decline in an indicator would be expected when an ecosystem becomes more impacted as a result of fishing, each of the 19 ecosystems was ranked between 1 (least impacted) and 19 (most impacted), to reflect weakest to strongest effects of fishing, based on the expected response of the indicator to fishing impact (Bundy *et al.*, 2010; Shin *et al.*, 2010a). For ranking according to the recent ecosystem states, the mean value of each indicator in the years 2003–2005

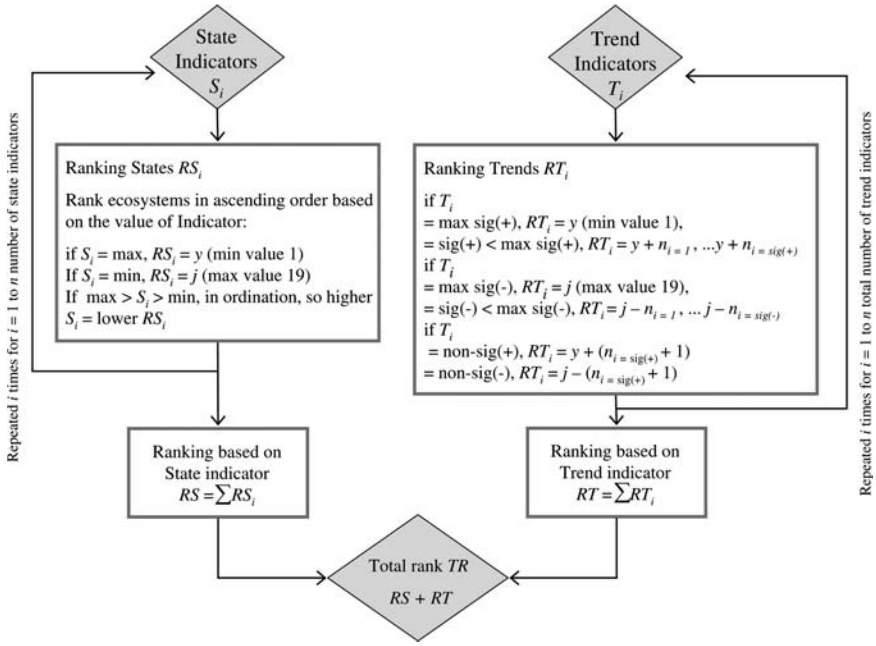

Figure 1. Schematic representation on the ranking process taking into account the indicators for states and for trends. The most-impacted ecosystem is ranked 19, the least is ranked 1.

was compared across ecosystems. Therefore, the ecosystem for which that indicator had the lowest value (highest impacted by fishing) was ranked 19 for that indicator and the following ecosystems were ranked accordingly (Figure 1). For overall ranking of trends, first the sign (whether it was positive or negative) and magnitude of the trend were considered, then the significance of the relationship (Figure 1). Therefore, when the trends were significantly negative, ecosystems were ranked according to how strongly (significantly steepest slope) they matched the expected declining trends. When the trends were not significant, the ecosystems were ranked using the sign of the trend, but each ecosystem having a negative trend for that indicator was given the same score (rank) without distinguishing between ecosystems, and each ecosystem for which the indicator changed in a positive but not significant direction over time was given the same (but lower) score. Indicators that had significant positive trends opposite to that expected with increasing fishing impacts were ranked lowest because they were showing the weakest effects of fishing.

When an indicator regarding states or trends was not available, the ecosystem was given a mean value of the ranking, so precluding underestimation of the ranking by counting missing values as zero. This was the case for Canadian west coast, Mauritania, Morocco (Sahara coastal), and the Bay of Biscay, where one indicator of state was missing, and for the Canadian west coast, Morocco, the Bay of Biscay, and the northern Humboldt, where one indicator for trends was also missing. Confounding and other pressures on the ecosystem were taken into account when discussing the results based on the knowledge of experts from each system and the results from abiotic indicators.

Using the set of selected indicators, the summed scores over all indicators were calculated so that an overall rank by ecosystem based on relative state, recent trends, and both states and trends was provided (Figure 1). These results were combined to classify the ecosystems as currently most, moderately, or least impacted and those becoming more, moderately, or less impacted by fishing. First and third quartiles were used to classify the ecosystems within least- and most-impacted categories. A weighted overall score summing partial scores was then calculated to rank ecosystems considering the four ecosystem attributes.

Comparing similarities between ecosystems

Similarities between ecosystems attributable to ecological indicators were then explored by principal component analysis (PCA; Jongman *et al*., 1999) using PRIMER (Clarke and Warwick, 2001). Data were normalized before comparisons to focus attention on patterns within the whole ecosystem and to take into account contributions from properties with different scales. PCA on the correlation matrix was used to reduce the number of multivariate dimensions to a smaller set of linear combinations that explained the most variance. This allowed the construction of a similarity map of the ecosystems.

Contrasting drivers of ecosystem response

Similarities between ecosystems regarding ecological and abiotic factors were further explored using non-parametric statistical procedures in PRIMER. The BIO–ENV routine (Clarke and Warwick, 2001) enabled us to identify the abiotic factors that globally best explained variability in the ecosystems attributable to the ecological indicators. This procedure calculated the correlation coefficients between similarity matrices of ecosystems from both ecological and abiotic indicators (described below), and identified the combination of abiotic factors that maximized the correlation between ecological and abiotic similarity matrices using indicators of both trends and states. Spearman's rank correlations were then applied to analyse the individual correlations between the ecological and abiotic indicators, as well as with the ranking results.

Abiotic indicators included in the analyses were a series of environmental and socio-economic factors (Table 1): (i) ecosystem type (temperate, tropical, upwelling, high latitude), (ii) primary productivity (mg C m^{-2} d^{-1}, mean value of 1998 from

Table 3. Ranking of the 19 ecosystems according to states and trends (based on Figures 2–4).

Ranking quartile	By state (2003–2005)	By short-term trend (1996–2005)	By long-term trend (1980–2005)	By state and short-term trend	By state and long-term trend
Fourth quartile (highest)	*N Humboldt*	**NC Adriatic Sea**	**C Baltic Sea**	**NC Adriatic Sea**	**C Baltic Sea**
	NC Adriatic Sea	*NEUS*	E Scotian shelf	*North Sea*	**S Catalan Sea**
	Bay of Biscay	*North Sea*	**S Catalan Sea**	*NEUS*	**NC Adriatic Sea**
	S Catalan Sea	*S Benguela*	Senegal	*S Benguela*	*N Humboldt*
	C Baltic Sea	*Mauritania*	–	*Mauritania*	–
Third quartile	Guinea	**Senegal**	**S Humboldt**	**Senegal**	E Scotian shelf
	S Humboldt	*Irish Sea*	NC Adriatic Sea	*C Baltic Sea*	**Senegal**
	Portugal	*C Baltic Sea*	N Humboldt	Morocco	**S Humboldt**
	–	**S Humboldt**	–	*Irish Sea*	–
	–	–	–	**S Humboldt**	–
Median	**Morocco**	–	S Benguela	–	Bay of Biscay
	Senegal	**Morocco**	**Morocco**	–	–
	–	–	Mauritania	–	–
Second quartile	*Mauritania*	*Bering Sea*	North Sea	**Guinea**	Morocco
	Barents Sea	**Barents Sea**	**Irish Sea**	*Bering Sea*	*Mauritania*
	Irish Sea	**Guinea**	NEUS	E Scotian shelf	**Guinea**
	S Benguela	S Catalan Sea	**Barents Sea**	**Barents Sea**	*S Benguela*
	–	–	–	–	**Irish Sea**
	–	–	–	–	Portugal
	–	–	–	–	**Barents Sea**
First quartile (lowest)	**Canada West C**	*N Humboldt*	Guinea	S Catalan Sea	*NEUS*
	NEUS	*E Scotian shelf*	**Bay of Biscay**	*N Humboldt*	*North Sea*
	E Scotian shelf	**Portugal**	**Bering Sea**	**Bay of Biscay**	**Canada West C**
	North Sea	**Bay of Biscay**	**Portugal**	**Canada West C**	**Bering Sea**
	Bering Sea	**Canada West C**	**Canada West C**	**Portugal**	–

Quartiles (based about the median score) are numbered four (for the highest quartile, containing the most-impacted ecosystems) to one (the lowest quartile with the least-impacted systems). Within each quartile, ecosystems are listed in descending order. Italicized entries denote ecosystems that are in the same quartile in two ranking cases, and emboldened entries denote ecosystems that are in the same quartile in three or more cases.

large marine ecosystems, www.lme.noaa.gov, in www.seaaroundus.org), (iii) annual mean sea surface temperature (SST, °C; Smith and Reynolds, 2004), (iv) fisheries type (mainly industrial or both artisanal and industrial fisheries), (v) enforcement of management regulations (low, medium, or high), both obtained from ecosystem experts reflected in Shin *et al*. (2010a), (vi) the human development index (HDI, UNDP, http://hdr.undp.org), and (vii) total landings (provided by experts within each ecosystem). Categorical indicators were transformed into ordinal variables classifying the ecosystems. For example, fishing type was classified as (i) when fishing only including industrial activities, and (ii) when including both industrial and artisanal fleets. Ecosystem type was classified geographically from the equator to the poles. A more detailed analysis of ecosystem drivers is performed by Link *et al*. (2010).

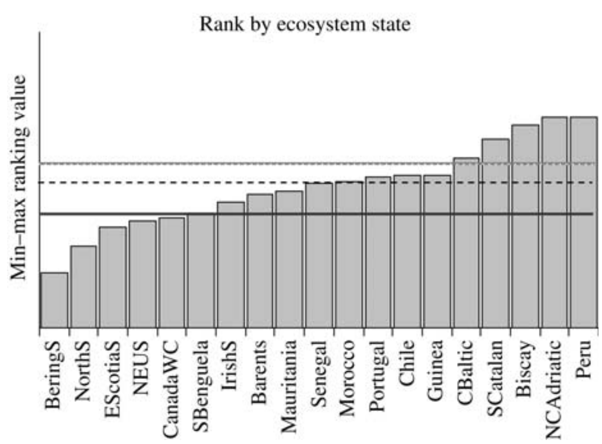

Figure 2. Overall ranking of the 19 ecosystems according to states (2003–2005). The dashed horizontal line denotes the median of the ranking values. The first quartile (black line) and third quartile (grey line) are indicated. Higher ranking indicates that an ecosystem is more strongly impacted by fishing. EScotiaS, eastern Scotian shelf; CanadaWC, Canadian west coast; NEUS, northeastern USA continental shelf; Chile, southern Humboldt; CBaltic, central Baltic Sea; SCatalan, southern Catalan Sea; NCAdriatic, north-central Adriatic Sea; Peru, northern Humboldt.

Results

Ranking ecosystems by state and trend

Ranking with short- and long-term trends and states differed notably because of the differences in the trends (Table 3). Although there was no ecosystem that rated in the most highly impacted quartile of systems for every case, the north-central Adriatic Sea, the southern Catalan Sea (both in the Mediterranean), and the central Baltic Sea did rank as highly impacted usually (Figures 2–4). For the last two of these ecosystems, the results were only consistent for cases involving states and long-term trends (Figures 2 and 3b).

This pattern of consistent ranking for just one type of trend or another (i.e. for short- or long-term trends, but not for both) was also seen for the northeastern US, the North Sea, the southern

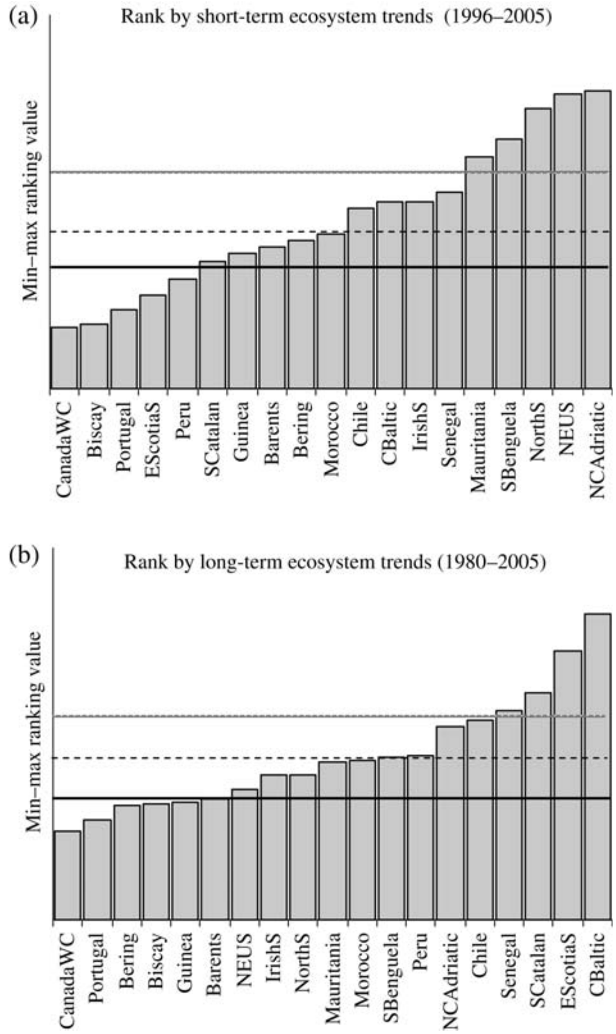

Figure 3. Overall ranking of the 19 ecosystems in terms of (a) short-term (1996 – 2005) and (b) long-term trends (1980 – 2005). The median score (dashed line), first quartile (black line), and third quartile (grey line) are indicated. Higher ranking indicates that an ecosystem is more strongly impacted by fishing.

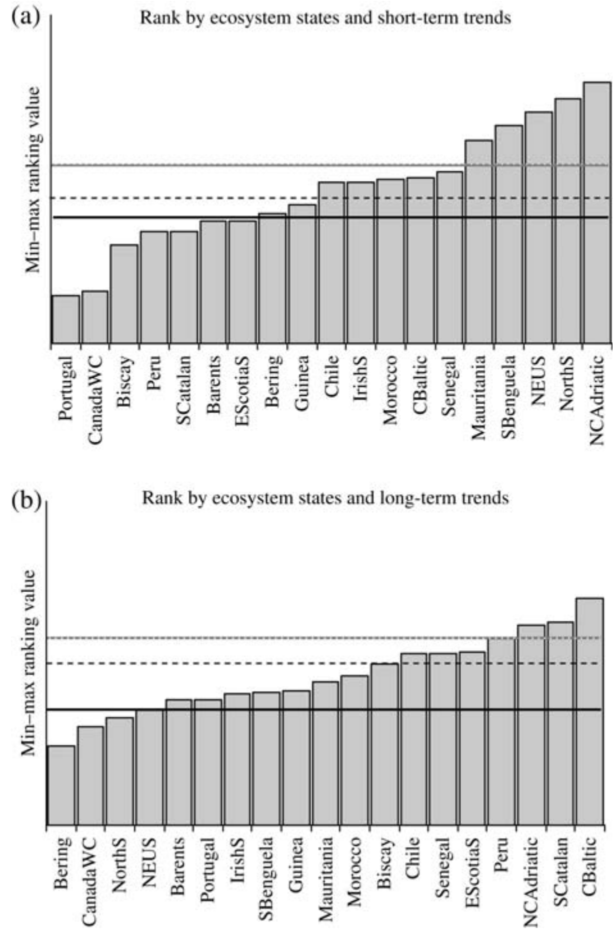

Figure 4. Overall combined ranking of the 19 ecosystems regarding both states (2003 – 2005) and trends considering (a) short-term (1996 – 2005) and (b) long-term trends (1980 – 2005). The median score (dashed line), the first quartile (black line), and the third quartile (grey line) are indicated. Higher ranking means more strongly impacted ecosystem by fishing.

Benguela, and Mauritania (heavily impacted if short-term trends were considered, but Mauritania has a median level of impact for long-term trends), the eastern Scotian Shelf (heavily impacted if long-term trends are considered), and the northern Humboldt (among the least impacted if short-term trends were considered and highly impacted according to long-term trends; Figures 2–4). Rankings were less affected by the trend type among many of the less heavily impacted systems. The most consistently ranking ecosystem with the least impact of fishing was the Canadian west coast, which was in the lowest quartile of ranks for all five rankings listed in Table 3. Guinea and the Irish Sea were among the systems less heavily impacted by fishing usually across both trend types (Figure 3a and b). The Barents and Bering Seas were similarly among the least-impacted sites, regardless of the length of trend considered (Figure 3a and b).

Ranking ecosystems in relation to ecosystem attributes

At all the levels of fisheries impact, a few systems rated consistently over roughly half the combinations of ecological trend or under just one or the other trend type (Table 4). It was only at the extremes of most and least impacted where ecosystems were ranked consistently for most rankings across ecosystem attributes and both trend types. The southern Benguela and the north-central Adriatic Sea were consistently the most heavily impacted ecosystems, whereas the Canadian west coast and the Barents Sea were the least impacted.

Classification of ecosystems

Combining states and short-term trends, we classified the systems into nine potential categories according to whether the systems were most, moderately, or least impacted, and whether they were becoming more impacted, less impacted, or remaining stationary (Figure 5a). The classification was carried out using the ranking results by quartiles (Table 3, Figures 2 and 3). In terms of short-term trends and states, the north-central Adriatic Sea ranked as most impacted by fishing, and it was becoming worse. That system, along with the central Baltic Sea and the southern Catalan Sea, ranked as highly impacted by fishing and showed moderate or high trends that reflected the impacts of fishing. Two systems were classified as moderately impacted and becoming more impacted: the southern Benguela and Mauritania. The North Sea and

Table 4. Ranking of the 19 ecosystems according to ecosystem attributes (data by ecological goals not shown).

Ranking quartile	State (2003–2005) and short-term trend (1996–2005)				State (2003–2005) and long-term trend (1980–2005)			
	By EF	**By SR**	**By CB**	**By RP**	**By EF**	**By SR**	**By CB**	**By RP**
Fourth quartile (highest)	Irish Sea	North Sea	**NC Adriatic Sea**	*Senegal*	*C Baltic Sea*	N Humboldt	Barents Sea	*Senegal*
	S Benguela	**NC Adriatic Sea**	*Senegal*	**NC Adriatic Sea**	S Humboldt	**S Benguela**	*S Catalan Sea*	**NC Adriatic Sea**
	NEUS	N Humboldt	*Morocco*	Mauritania	*Morocco*	*S Catalan Sea*	*C Baltic Sea*	Guinea
	North Sea	**S Benguela**	Bering Sea	–	*S Catalan Sea*	*C Baltic Sea*	**S Benguela**	Bering Sea
	Mauritania	NEUS	**S Benguela**	–	**NC Adriatic Sea**	E Scotian shelf	*Morocco*	*C Baltic Sea*
Third quartile	Morocco	Bay of Biscay	S Humboldt	*NEUS*	*E Scotian shelf*	Irish Sea	*E Scotian shelf*	Mauritania
	NC Adriatic Sea	–	Barents Sea	Guinea	Irish Sea	Guinea	*NEUS*	S Humboldt
	E Scotian shelf	–	C Baltic Sea	Bering Sea	North Sea	Portugal	Bay of Biscay	–
	S Catalan Sea	–	*NEUS*	C Baltic Sea	–	Mauritania	NC Adriatic Sea	–
Median	Guinea	–	North Sea	–	–	NEUS	Senegal	Portugal
	Barents Sea	C Baltic Sea	–	Portugal	Senegal	Morocco	–	N Humboldt
	C Baltic Sea	–	–	–	–	NC Adriatic Sea	–	–
Second quartile	*S Humboldt*	Senegal	*E Scotian shelf*	Irish Sea	*Mauritania*	Canada West Coast	*N Humboldt*	*E Scotian shelf*
	N Humboldt	*E Scotian Shelf*	*Mauritania*	*S Humboldt*	*N Humboldt*	Senegal	*S Humboldt*	*Bay of Biscay*
	–	*Mauritania*	Guinea	*S Catalan Sea*	Portugal	North Sea	Bering Sea	NEUS
	–	*S Humboldt*	*Bay of Biscay*	North Sea	*Bay of Biscay*	–	–	*S Catalan Sea*
	–	*S Catalan Sea*	–	–	–	–	–	–
	–	Guinea	–	–	–	–	–	–
	–	Irish Sea	–	–	–	–	–	–
First quartile (lowest)	**Canada West Coast**	*Portugal*	N Humboldt	E Scotian Shelf	Guinea	S Humboldt	*Portugal*	**Canada West Coast**
	Senegal	**Barents Sea**	*Irish Sea*	*Bay of Biscay*	*S Benguela*	*Bay of Biscay*	Guinea	**Barents Sea**
	Bering Sea	**Canada West Coast**	**Canada West Coast**	**Barents Sea**	**Canada West Coast**	**Barents Sea**	**Canada West Coast**	*Irish Sea*
	Bay of Biscay	*Morocco*	*Portugal*	**Canada West Coast**	**Barents Sea**	*Bering Sea*	Mauritania	*North Sea*
	Portugal	–	S Catalan Sea	N Humboldt	NEUS	–	*Irish Sea*	*S Benguela*
	–	–	–	*S Benguela*	*Bering Sea*	–	*North Sea*	*Morocco*
	–	–	–	*Morocco*	–	–	–	–

Quartiles (based about the median score) are numbered four (for the highest quartile, containing the most-impacted ecosystems) to one (the lowest quartile with the least-impacted systems). Within each quartile, ecosystems are listed in descending order. Italicized entries denote ecosystems that are in the same quartile in 3–4 ranking cases, emboldened entries ecosystems that are in the same quartile in five or more cases, and underlined entries ecosystems that are in the same quartile for two ecosystem attributes within the same state/trend combination. By cases we mean states, short- and long-term trends, composites (state and trend), and ecological goal. EF, ecosystem structure and functioning; SR, ecosystem stability and resistance to perturbations; CB, conservation of functional biodiversity; RP, resource potential.

northeastern US continental shelf were classified as relatively less impacted in recent years (states) but becoming more impacted with time from 1996 to 2005. All seven ecosystems were classified as more strongly impacted by fishing. In contrast, the Bering Sea and eastern Scotian Shelf were classified as relatively less impacted by fishing, and the trends showed moderate to less impact. The Canadian west coast and Portugal were moderately impacted (recent state), showing trends towards lessening impacts of fishing. The northern Humboldt and the Bay of Biscay were most impacted, but were becoming less impacted. These six systems fell within the category of being less to moderately impacted by fishing. The classification of Morocco, Guinea, Senegal, the southern Humboldt, the Irish Sea, and the Barents Sea was considered unclear or intermediate (meaning that they were classified as moderately impacted by fishing and becoming moderately more impacted).

Although the position of the ecosystems in Figure 5b taking into account long-term trends did not exactly match that shown in Figure 5a, the overall classifications into less to moderately impacted, more strongly impacted, and unclear and needing close monitoring with long-term trends were largely unchanged from those with short-term trends. There were a few switches, however. In particular, Senegal and the southern Humboldt shifted from higher impacts to unclear, changing positions with the southern Benguela and Mauritania, whereas the

(a)

State / Trends	Most impacted	Moderately impacted	Less impacted
Becoming more impacted	NCAdriaticSea	SBenguela, Mauritania	NorthSea, NEUS
Moderately impacted	CBalticSea, SCatalanSea	Morocco, Chile, Guinea, IrishSea, Senegal, BarentsSea	BeringSea
Becoming less impacted	Peru, BayBiscay	CanadaWC, Portugal	EScotianShelf

(b)

State / Trends	Most impacted	Moderately impacted	Less impacted
Becoming more impacted	CBalticSea, SCatalanSea	Senegal, Chile	EScotianShelf
Moderately impacted	BayBiscay, NCAdriaticSea, Peru	SBenguela, IrishSea, BarentsSea, Mauritania, Morocco	NorthSea, NEUS
Becoming less impacted		CanadaWC, Portugal, Guinea	BeringSea

Unclear
Higher impacted
Lower-medium impacted

Figure 5. Diagnostic diagram classifying the 19 exploited ecosystems according to whether they are most, moderately, or least impacted by fishing (state analysis) and whether they are becoming more, moderately, or less impacted by fishing (trend results); includes trend data (a) from 1996 to 2005 and (b) from 1980 to 2005.

eastern Scotian Shelf, the Bay of Biscay, and the northern Humboldt dropped from higher to lesser impacts, changing positions with the North Sea and the northeastern US (Table 5).

Similarities between ecosystems

In terms of ecosystem states, the first two principal components (PCs) explained 86% of the variability in the data when performing a PCA (Figure 6). The first principal component (PC1) was explained by four factors: low sustainability of stocks, long lifespan, low fish size, and low percentage of predators. The northern Humboldt, the north-central Adriatic Sea, the North Sea, the southern Catalan Sea, the central Baltic Sea, and the southern Humboldt showed the lowest scores for PC1. The Bering Sea, the Canadian west coast, the northeastern US continental shelf, the eastern Scotian Shelf, the Barents Sea, and Portugal scored highest. PC2 was explained by low biomass stability, fish size, percentage of healthy stocks, and long lifespan. For this component, the northeastern US continental shelf stood out clearly, followed by the Canadian west coast, Portugal, the northern Humboldt, and the Bay of Biscay, which all scored similarly high values. The Bering Sea and the North Sea clearly had the lowest

Table 5. Summary of classification of ecosystems according to short- and long-term trends (see Figure 5 for detailed classifications).

Ecosystem	Situation
Barents Sea	U
Bay of Biscay	H → L
Bering Sea	L
Central Baltic Sea	H
Eastern Scotian shelf	H → L
Guinean EEZ	L → U
Irish Sea	U
Mauritania	U → H
Morocco	U
North Central Adriatic Sea	H
Northeastern US	L → H
North Sea	L → H
Northern Humboldt	H → L
Portugal EEZ	L
Senegal EEZ	H → U
Southern Benguela	U → H
Southern Catalan Sea	H
Southern Humboldt	H → U
West coast Canada	L

U, an unclear situation; L, lower impacts of fishing, i.e. a system classified as in a more lightly impacted by fishing situation; H, higher impacts, i.e. a more strongly impacted situation; → indicates that a system's classification changed when assessed over the long term (1980–2005, or for as long as there were data available for the system—see Link *et al*., 2010, for lengths of the trends) relative to the short term (1996–2005).

scores. Similarities were shown between ecosystem pairs (e.g. the northern Humboldt and Mediterranean case studies, the northeastern US continental shelf, and the Canadian west coast). The North Sea and the Bering Sea showed unique features. PCA results were overall in line with the ranking results for states (Figure 2).

In terms of short-term trends, PC1 and PC2 explained 57% of the variability in the data (Figure 7a), PC1 being explained by high values for lifespan, percentage of predators, trophic level, and fish size and low values of biomass, and PC2 by low biomass, low inverse fishing pressure, and high trophic level of the catch and percentage of predators. The northern Humboldt, the north-central Adriatic Sea, and Portugal scored mainly negatively for both factors, whereas the Irish Sea, the Bering Sea, the eastern Scotian shelf, and the southern Benguela scored mainly positively. For long-term trends, two components explained 58% of the variability between ecosystems when performing a PCA (Figure 7b), PC1 being explained by high values of all indicators except biomass, and PC2 by low values for biomass, inverse fishing pressure, fish size, and lifespan and high trophic level and percentage of predators. Similar results were found for the analysis with short-term trends, although the Irish Sea and the North Sea showed important changes over time.

Contrasting drivers of ecosystem response

The correlation between the state ecological indicators and abiotic data showed an overall low but significant correlation (BIO–ENV test, correlation coefficient $\rho = 0.285$, $p = 0.048$). The similarity matrices obtained with fisheries type and with enforcement correlated the highest with the similarity matrix of ecological indicators ($\rho = 0.285$ for both). The correlation between the short-term trend ecological indicators and abiotic factors showed an overall

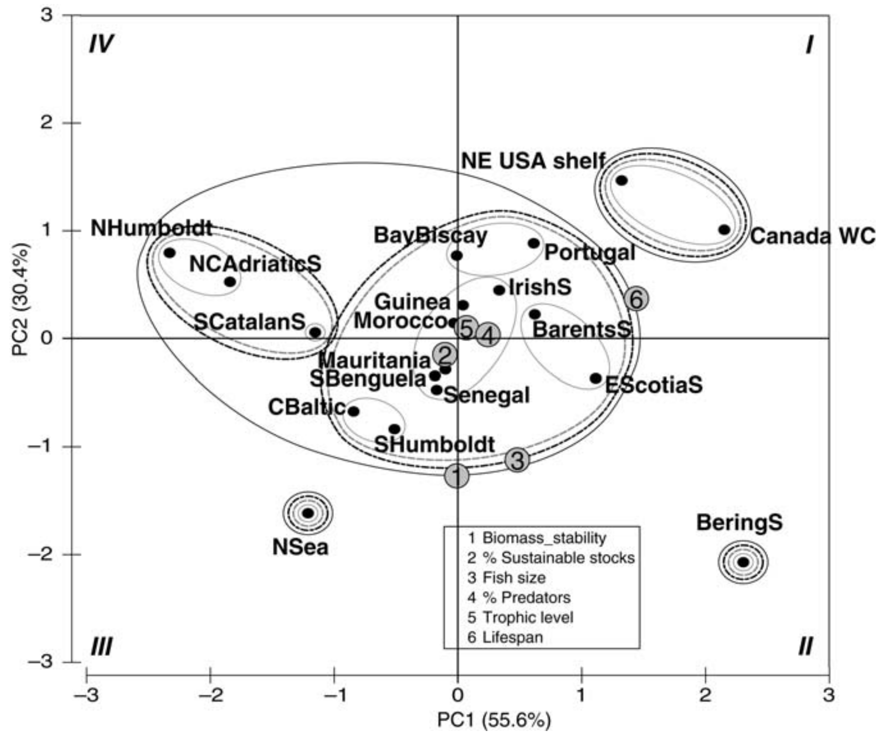

Figure 6. The first two principal components of the 19 exploited ecosystems and the six ecological indicators of state. Encircling lines represent decreasing Euclidean distances.

higher and significant correlation (BIO–ENV test, $\rho = 0.448$, $p = 0.01$). A combination of three factors (ecosystem type, fisheries type, and landed catches) correlated highest with the ecological indicators ($\rho = 0.480$). The correlation with the long-term trend ecological indicators also showed an overall significant correlation ($\rho = 0.376$, $p = 0.01$), and a combination of the same three factors (ecosystem type, fisheries type, and total landings) correlated highest with the ecological indicators ($\rho = 0.376$).

The correlation between the state ranking similarity matrix and the ecological similarity matrix showed high correlation (BIO–ENV test, $\rho = 0.439$, $p = 0.01$). The correlation between the similarity matrix of the ranking of the short- and long-term trend indicators and the ecological similarity matrix showed lower but still significant correlations (BIO–ENV test, $\rho = 0.063$, $p = 0.01$ and $\rho = 0.094$, $p = 0.01$, respectively). Therefore, the ranking of ecosystems was a better way of summarizing the information from ecological indicators for states, then for long-term trends, and finally for short-term trends.

Logically, the ranking for state indicators was negatively correlated with all ecological indicators using the Spearman rank correlation (Table 6). Similarly, most negative correlations were between ranking from long-term trends and ecological indicators (Table 7). However, the results looking at rankings of short-term trends showed significant positive correlations with the trophic level (Table 7). Contrary to expectations, that ecological indicator showed an increase when ecosystems were ranked overall as more strongly impacted by fishing.

When correlating the ranks of states and trends (Table 8), rankings in terms of states and long-term trends were positively correlated (although not significantly), which may indicate that when an ecosystem is highly impacted by fishing now, high impacts of fishing are also found in long-term trends. However, short-term trends may not show similar evidence, underscoring the need for both trends and state indicators to be used to categorize ecosystems, and suggesting that longer time-series of data should be used wherever possible because it is more difficult to detect significant clear trends using shorter time-series. Further exploration of correlations for ecological and abiotic indicators from states and trends showed high correlations between the two (Tables 6 and 7).

In terms of the correlation with abiotic indicators, the ranking resulting from state indicators correlated positively with fisheries type (the ranking was higher when there were both industrial and artisanal fisheries) and with SST, and negatively correlated with enforcement (higher ranking when enforcement was lower; Table 6). Ranking results from short-term trends were negatively correlated with ecosystem type (higher ranks or impacts of fishing in temperate systems), and positively correlated with higher primary production and higher SST (Table 7). The ranking results from long-term trends were negatively correlated with ecosystem type (suggesting a higher impact of fishing in temperate areas), with enforcement (higher impacts in low-enforcement situations), and with catches (higher impacts with lower catches, perhaps illustrating the depletion of ecosystems), but positively correlated with primary production, SST, and HDI (Table 7).

Discussion

Rank and classification of marine ecosystems regarding fishing impacts

Our ranking exercise has highlighted the importance of analysing both states and trends in assessing the impacts of fishing on marine ecosystems. Such ranking, particularly using states and short-term trends, produced different results. Insights into the reasons may be

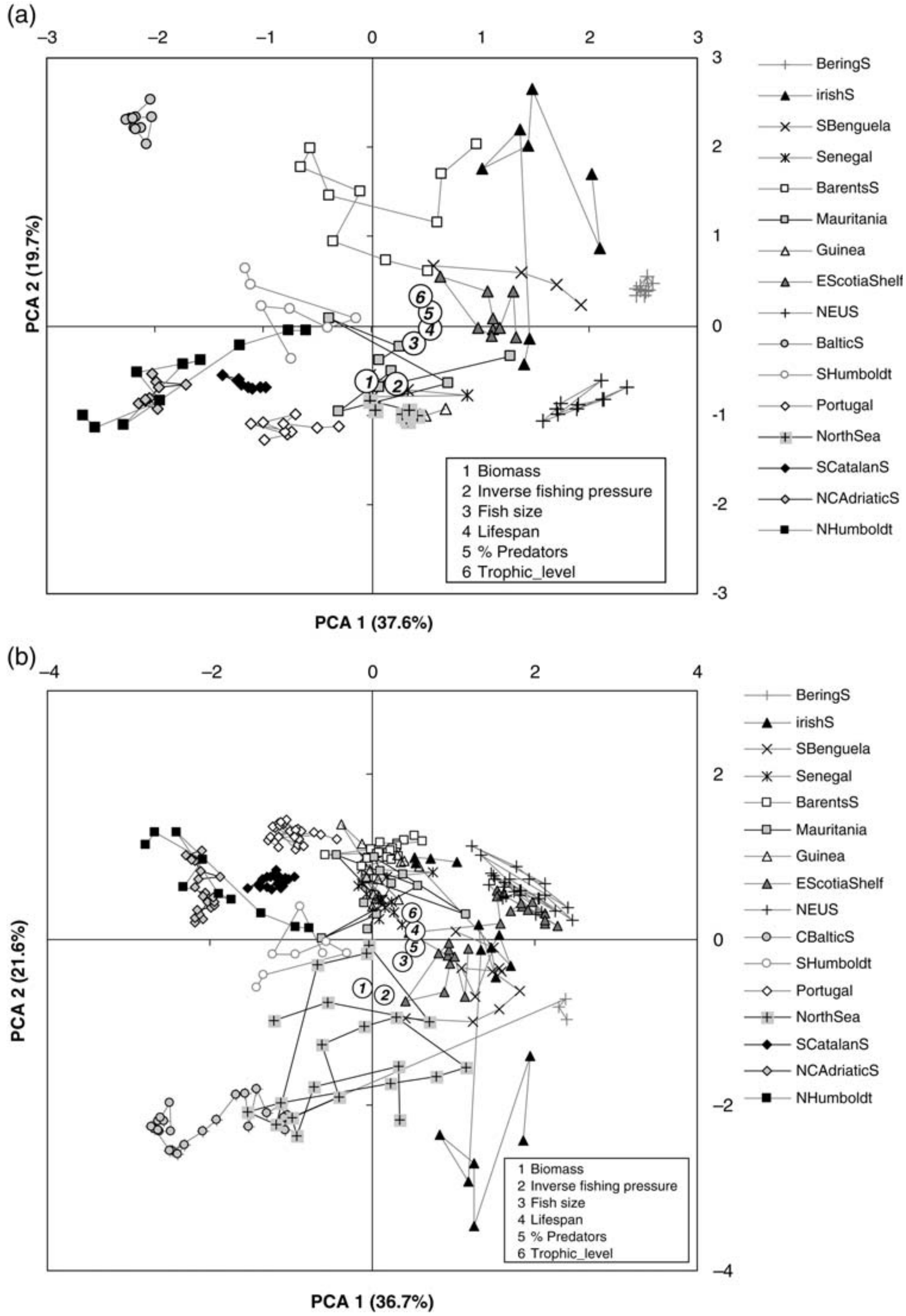

Figure 7. The first two principal components of the 19 exploited ecosystems and the six ecological indicators of trends for the periods (a) 1996–2005 and (b) 1980–2005.

drawn from the overlap of ecosystems that ranked high (more impacted) according to states and long-term trends, and ecosystems that ranked low (less impacted) according to short- and long-term trends. This is consistent with what was observed when analysing the PCs of ecological indicators. Clearly, the length of the time-series is important, because ecosystems change over time and depend on the ecological and exploitation history, environmental variations, and such other factors as pollution and eutrophication. Short- and long-term trends may provide different outcomes from the analysis, and short-term trends may fail to capture the main trends (Blanchard *et al.*, 2010).

Ranking each ecosystem according to the four ecosystem attributes demonstrated the need to use a combination of indicators addressing different ecosystem attributes to characterize exploited ecosystems and the progress towards an EAF. This confirms the results of earlier studies which concluded that proper characterization of ecosystem status requires multiple indicators (Murawski, 2000; Fulton *et al.*, 2005; Link, 2005). No ecosystem

Table 6. Results of Spearman's rank correlations between the ecological indicators of states (2003–2005), abiotic indicators (Table 2), and the ranking of states (Figure 2) from the 19 exploited ecosystems.

	Environmental and socio-economic indicators							Ecological indicators of state (2003–2005)					
Spearman's ρ ($n = 19$)	**Ecotype[a]**	**Fisheries[b]**	**Enforcement[c]**	**HDI[d]**	**PP[e]**	**SST[f]**	**Catches[g]**	**Fish size**	**Lifespan**	**% Predators**	**Biomass stability**	**Trophic level**	**% Healthy stocks**
Ecotype[a]	–												
Fisheries[b]	0.21	–											
Enforcement[c]	−0.18	**−0.84**	–										
HDI[d]	**−0.64**	**−0.68**	**0.59**	–									
PP[e]	−0.02	−0.21	0.11	−0.16	–								
SST[f]	0.07	**0.82**	**−0.76**	**−0.68**	0.08	–							
Catches[g]	***0.48***	−0.23	0.13	−0.20	0.18	−0.24	–						
Fish size	0.37	−0.18	0.24	−0.24	***0.45***	−0.16	0.03	–					
Lifespan	−0.05	***−0.49***	***0.47***	0.32	0.03	−0.38	0.20	0.15	–				
% Predators	−0.16	***−0.53***	0.41	0.24	0.34	−0.02	−0.04	0.22	***0.51***	–			
Biomass stability	0.27	−0.14	0.11	−0.05	0.03	−0.31	−0.15	0.45	−0.32	−0.24	–		
Trophic level	0.00	***−0.47***	0.44	0.41	−0.17	−0.34	0.18	−0.09	***0.48***	0.43	−0.33	–	
% Healthy stocks	0.01	−0.02	0.24	−0.09	0.44	−0.03	−0.06	0.34	−0.17	−0.06	0.19	−0.29	–
State rank[h]	−0.07	**0.68**	**−0.67**	−0.33	−0.37	***0.45***	−0.05	**−0.58**	**−0.60**	**−0.70**	−0.17	***−0.52***	−0.27

Emboldened and italicized, correlation significant at the 0.05 level (two-tailed); emboldened, correlation is significant at the 0.01 level (two-tailed).
[a]Ecosystem type: 1, temperate; 2, tropical; 3, upwelling; 4, high latitude.
[b]Fisheries type: 1, industrial; 2, industrial and artisanal.
[c]Fisheries enforcement: 1, low; 2, medium; 3, high.
[d]Human development index.
[e]Primary productivity (mg C m^{-2} d^{-1}).
[f]Sea surface temperature (°C).
[g]Total official landings (t).
[h]Rank obtained analysing state data (2003–2005).

Table 7. Results of Spearman's rank correlations between the ecological indicators of trends, abiotic indicators (Table 2), and the ranking of trends (Figure 3) from the 19 exploited ecosystems.

	Environmental and socio-economic indicators							Ecological indicators of state (2003–2005)					
Spearman's ρ	**EcoType[a]**	**Fisheries[b]**	**Enforcement[c]**	**HDI[d]**	**PP[e]**	**SST[f]**	**Catches[g]**	**Fish size**	**Lifespan**	**% Predators**	**Biomass**	**Trophic level**	**Inv. fishing pressure**
$n = 129$													
Ecotype[a]	–												
Fisheries[b]	**0.19**	–											
Enforcement[c]	−0.13	***−0.88***	–										
HDI[d]	***−0.58***	***−0.67***	***0.60***	–									
PP[e]	***−0.34***	***−0.27***	**0.18**	−0.02	–								
SST[f]	***−0.20***	***0.68***	***−0.71***	***−0.62***	***0.31***	–							
Catches[g]	***0.47***	−0.12	0.07	***−0.27***	0.14	−0.04	–						
Fish size	***0.23***	***−0.27***	**0.38**	−0.12	**0.25**	**−0.20**	−0.13	–					
Lifespan	***0.21***	***−0.41***	***0.43***	0.06	***0.34***	***−0.28***	0.05	***0.36***	–				
% Predators	***−0.17***	***−0.42***	***0.43***	***0.32***	***0.31***	***−0.24***	***−0.25***	***0.30***	***0.53***	–			
Biomass	***0.32***	***−0.47***	***0.39***	0.12	0.04	***−0.39***	***0.82***	−0.01	0.12	−0.06	–		
Trophic level	***0.29***	***−0.36***	***0.53***	0.14	−0.12	***−0.51***	***0.21***	**0.17**	***0.40***	***0.43***	***0.27***	–	
Inv. fishing pressure	0.15	***−0.57***	***0.52***	***0.43***	−0.10	***−0.67***	***0.16***	***0.28***	***0.31***	0.11	***0.61***	**0.21**	–
Short-term trends rank	***−0.37***	−0.01	0.16	−0.05	***0.23***	**0.18**	0.02	0.02	−0.08	0.14	−0.03	**0.20**	**−0.22**
$n = 299$													
Ecotype[a]	–												
Fisheries[b]	0.05	–											
Enforcement[c]	−0.10	***−0.83***	–										
HDI[d]	***−0.69***	***−0.61***	***0.59***	–									
PP[e]	***−0.15***	***−0.42***	***0.24***	**0.15**	–								
SST[f]	***−0.17***	***0.57***	***−0.65***	***−0.48***	**0.15**	–							
Catches[g]	***0.45***	***−0.18***	0.10	***−0.30***	**0.16**	0.06	–						
Fish size	***0.27***	***−0.39***	***0.36***	−0.11	***0.50***	**−0.15**	−0.07	–					
Lifespan	***0.19***	***−0.35***	***0.47***	0.06	***0.25***	***−0.51***	***0.18***	***0.35***	–				
% Predators	0.00	***−0.37***	***0.28***	***0.25***	***0.49***	***−0.21***	**−0.16**	***0.28***	***0.48***	–			
Biomass	***0.40***	***−0.46***	***0.19***	−0.07	***0.23***	−0.10	***0.80***	0.08	***0.20***	0.05	–		
Trophic level	***0.20***	***−0.38***	***0.53***	***0.26***	0.11	***−0.56***	0.08	***0.25***	***0.48***	***0.51***	***0.17***	–	
Inv. fishing pressure	***0.35***	***−0.40***	***0.17***	−0.06	0.11	***−0.27***	***0.29***	***0.24***	***0.23***	0.02	***0.66***	**0.12**	–
Long-term trends rank[h]	***−0.37***	0.01	***−0.36***	***0.13***	***0.38***	***0.41***	***−0.18***	0.10	***−0.45***	***−0.12***	0.07	***−0.54***	**0.11**

Results from (top panel) short-time trends (1996–2005), and (bottom panel) long-term trends (1980–2005). Emboldened, correlation significant at the 0.05 level (two-tailed); emboldened and italicized, correlation significant at the 0.01 level (two-tailed).

[a]Ecosystem type: 1, temperate; 2, tropical; 3, upwelling; 4, high latitude.

[b]Fisheries type: 1, industrial; 2, industrial and artisanal.

[c]Fisheries enforcement: 1, low; 2, medium; 3, high.

[d]Human development index.

[e]Primary productivity (mg C m^{-2} d^{-1}).

[f]Sea surface temperature (°C).

[g]Total official landings (t).

[h]Rank obtained analysing time-series of data, 1996–2005 (top panel) and 1980–2005 (bottom panel).

Table 8. Results of Spearman's rank correlations between the ranking of states and trends (Figures 2–4) from the 19 exploited ecosystems.

Spearman's ρ (n = 19)	State rank	Long-term trends rank	Short-term trends rank	All ranking[a]	All ranking[b]
State rank	–				
Long-term trends rank	0.31	–			
Short-term trends rank	−0.17	0.33	–		
All ranking[a]	**0.74**	**0.83**	0.01	–	
All ranking[b]	−0.18	0.36	**0.97**	0.05	–

Emboldened correlation is significant at the 0.01 level (two-tailed).
[a]Includes long-term trends, 1980–2005.
[b]Includes short-term trends, 1996–2005.

showed higher or lower ranks for all ecosystem attributes for short-term trends, and only one ecosystem, the eastern Scotian Shelf, showed higher ranks for all four ecosystem attributes when considering long-term trends. Therefore, if a single ecological indicator had been selected, different conclusions may have been drawn. For the North Sea, our results show how fishing has impacted the SR and EF during the period 1996–2005. However, the results suggest intermediate impacts when considering only CB as an indicator, and less impact of fishing was shown when looking at RP. Ecosystem attributes in terms of maintaining the total biomass of surveyed species and controlling fishing effort are likely being achieved in the North Sea, but the system is still highly impacted if other aspects of ecosystem overfishing are considered (Murawski, 2000). Therefore, although fishing mortality and exploitation rate may have declined in the North Sea in the past decade (see also Daan *et al*., 2005), trends in the other indicators have not shown an improvement, likely reflecting the delayed response of ecosystems in recovering from exploitation pressure.

Our diagnostic graphs indicated that of the systems that could be classified, more fell in the category of highly impacted by fishing according to long-term trends (42%) combined with recent states. The proportion of ecosystems classified as unclear/intermediately impacted increased in recent years (from 26 to 32%), whereas the proportion of ecosystems classified as being in moderate- or less-impacted states was maintained (32%). A significant fraction of systems is not clearly classified, supporting calls for early adoption of the precautionary approach when managing marine ecosystems (FAO, 1996).

Emergent patterns of fishing impacts

Three sets emerged when ranking our 19 ecosystems in terms of fishing impact, and these are outline below.

(i) *Intermediate impacts of fishing*. A group of ecosystems (Portugal, Barents Sea, Canadian west coast, Morocco, the Irish Sea, the southern Humboldt, and the Bering Sea) currently rank as intermediate or more lightly impacted and also have short- and long-term trends that do not change much over time. This pattern of intermediate fishing impacts for some ecosystems warrants careful interpretation because this work expresses fishing impacts in relative terms; lower rankings do not necessarily imply low impacts of fishing, only lower compared with the rest of the ecosystems analysed. Nevertheless, intermediate impacts of fishing on these ecosystems are reasonable results if the information available from these systems is considered (e.g. Loughlin and Ohtani, 1999; Coll *et al*., 2006a; Boldt, 2009; Shannon *et al*., 2009a).

(ii) *Higher impacts of fishing*. A second group consists of ecosystems showing higher impacts of fishing (the African case studies of Mauritania, Guinea, Senegal, and the southern Benguela, as well as the north-central Adriatic Sea, the North Sea, and the northeastern US continental shelf). Those systems have become more impacted in the past 10 years, so their state is moving to one that is more strongly impacted. This conclusion is mainly in line with information documented for these systems (e.g. Fogarty and Murawski, 1998; Jouffre and Domain, 1999; Link and Brodziak, 2002; Lobry *et al*., 2003; Domalain *et al*., 2004; Jouffre *et al*., 2004a, b; Lotze *et al*., 2006; Coll *et al*., 2007, 2009; Gascuel *et al*., 2007; Shannon *et al*., 2010), although the southern Benguela is only showing signs of heavier impacts by fishing in recent years, potentially because of natural fluctuations in the abundance of forage fish recently (Shannon *et al*., 2009a). The results for the African case studies are grouped together in the various analyses presented here, as one would expect from both their geographic position and their respective exploitation history (Chavance *et al*., 2004). At a regional scale, the environmental and fisheries characteristics certainly (relatively) contrast between the countries prosecuting the fisheries (Cury and Roy, 1991; Domain *et al*., 1999; Chavance *et al*., 2004). Nevertheless, these contrasts are less significant on average than those observable among the 19 ecosystems considered here. The North Sea, for instance, was more impacted from 1996 to 2005, so fishing likely was an important driver for the situation, as stated by Mackinson *et al*. (2009). However, it was in a lower ranking (i.e. relatively less impacted by fishing) from 1980 to 2005. These results are in line with trends in fishing effort, which declined for pelagic species in the North Sea up to the 1970s but increased again in the 1980s, whereas effort targeting demersal species was maintained or increased (Greenstreet *et al*., 1999). When looking at its current state (2003–2005), the North Sea ranked as less impacted (also see Daan *et al*., 2005). However, the results may also indicate that other drivers, such as climate, are influencing the North Sea too (Beaugrand *et al*., 2003; Heath, 2005). Evidence of changes in the zooplankton have pointed to regime shifts in 1988/1989 and 1998 (Richardson, 2008), so the changes in plankton and continued declines in some fish stocks suggest that the changes captured by our indicators for the North Sea cannot solely be attributed to the effects of fishing.

(iii) *Highly impacted but showing less-impacted states*. The last group of ecosystems (southern Catalan Sea, the eastern Scotian Shelf, the central Baltic Sea, the northern Humboldt, and the Bay of Biscay) shows long-term degradation, but short-term trends are towards a less-impacted state. However, they all show low rankings for short-term trends because of the low ranks for lifespan, percentage of predators, and/or mean trophic level of the catch (i.e.

higher values were obtained for these indicators). Information available for these five ecosystems suggests that the indicators may be failing to capture the impacts of fishing because of changes in the ecosystem and specific features of fishing. In the southern Catalan Sea, increased fishing effort on small pelagic fish (mainly on anchovy) and climate impacts (mainly impacting sardine) has led to a progressive decline in forage fish in the ecosystem (Palomera *et al*., 2007; Coll *et al*., 2008a). As small pelagic fish are small in size, have restricted lifespans, are located at low trophic levels, and are not predators, when they decrease in the system, the result would be an increase in lifespan, mean trophic level, and percentage of predatory fish ratio indicators. Moreover, because of the importance of their biomass in that system (Coll *et al*., 2006b), the resulting decrease in biomass also caused an increase in the inverse fishing effort indicator. Therefore, these indicators would be highlighting changes in the ecosystem attributable to changes in the small pelagic fish communities, which could make simple indicators more difficult to interpret (Shannon *et al*., 2009a). The situation may be similar when there is intensive exploitation of benthic invertebrates, as is true on the eastern Scotian Shelf (Bundy, 2005; Frank *et al*., 2005; Anderson *et al*., 2008). Moreover, the over-exploited state of some of these systems, e.g. the southern Catalan Sea and the eastern Scotian Shelf, may imply that they are no longer as responsive to further fishing effects as would be less-impacted systems (Bundy, 2005; Shannon *et al*., 2009a). Similar complexities prevail in the southern and northern Benguela upwelling systems (Cury and Shannon, 2004; Heymans *et al*., 2004; Cury *et al*., 2005; Watermeyer *et al*., 2008a, b; Shannon *et al*., 2009a, b). Upwelling ecosystems are driven mainly by the environmental factors that strongly influence some of the indicators we used to rank the ecosystems. The abundance of small pelagic fish characterizes these ecosystems and partly dominates their dynamics (Alheit and Ñiquen, 2004; Ballón *et al*., 2008; Chavez *et al*., 2008; Guevara-Carrasco and Lleonart, 2008; Shannon *et al*., 2008; Taylor *et al*., 2008). For example, the ranking of the northern Humboldt as more strongly impacted for recent states and long-term trends is likely due to the combination of environmental and fishing impacts. When ranking that ecosystem according to recent trends, the ranking improves, likely indicating changes in the environment and/or in fisheries management. Further discussion of upwelling or similar systems is provided by Shannon *et al*. (2010) and Bundy *et al*. (2010). In terms of the central Baltic Sea, lower values of indicators for mean lifespan, mean length, and mean trophic level of the catch may be attributable to a regime shift during the late 1980s, when the cod-dominated system there was replaced by one dominated by clupeids. The difference in the performance of the central Baltic Sea ecosystem in the long and short term (and therefore, its relative status) is probably also affected by the two contrasting ecosystem regimes over the long term, whereas the shorter period covers just one regime. The central Baltic Sea is characterized by a community showing the domination of pelagic fish (herring and sprat) over demersal (mainly cod-dominated; Möllmann *et al*., 2009).

In summary, our results on the ranking of states and trends clearly reflect differing changes, processes, and dynamics in each ecosystem, again showing that to fully understand what is happening in an ecosystem, it is important to include analyses of both states and trends. It is also fundamental to have information on the ecological, environmental, and fishery histories to be able to interpret the trends in indicators correctly and to disentangle the effects of fishing and of the environment, especially when lower-trophic-level species are heavily fished and when small pelagic fish are abundant.

Drivers of ecosystem responses

Multivariate analyses have helped to understand the differences and similarities between ecosystems, and in parallel with the rankings, to elucidate emergent patterns and explore complementary drivers of the patterns. PCA plots have suggested similar groupings to those based on a synthesis of the ranking of ecosystems according to states and trends. For example, in Figures 6 and 7, the north-central Adriatic and northern Humboldt group together, as they do in Figures 2 and 3b (where they were classified as heavily impacted and becoming more impacted). The southern Catalan Sea, the central Baltic Sea, and the southern Humboldt all fell into one of the four highly impacted categories (Figure 5b), and they grouped together in Figure 6. These comparisons of simple ranking-derived categorizations with aggregations of ecosystems by multivariate techniques may indicate cause for concern. Does the PCA grouping for the ecosystem state of Portugal with the Bay of Biscay, or of the northeastern US continental shelf with the Canadian west coast, perhaps suggest some warning signals that Portugal and the Canadian west coast may be heading for problems although their global combined classification of trends and states is showing improvement over time? Or perhaps are the northeastern US continental shelf and the Bay of Biscay showing a trend of recovery from a poorer state? The results may also highlight similarities in ecosystem structure. Nevertheless, our results do show some worrying trends for African case studies: they are ranked as moderately impacted by fishing in current states, but some trends (e.g. for Senegal and Mauritania) are showing signs of progression towards depletion.

The results also demonstrated that differences and similarities between the ecosystems attributable to ecological indicators and the resulting ranking partially correlate with abiotic factors. Ecosystem type is an important factor, and temperate systems are clearly more heavily impacted by fishing than high-latitude systems (suggesting geographic expansion of fishing). The enforcement of fisheries negatively correlated with the rankings of states and long-term trends, showing that ecosystems with poorer enforcement are the ones with greater fishing impacts. When looking at trends, high primary production was correlated with enhanced impacts of fishing (the more productive the ecosystem, the more impacted it was by fishing historically), and in both state and trend rankings, the warmer the system, the more impacted it was by fishing. This may reflect some ecological implications of climate change. Given the discussions about access and social objectives, it is interesting that when looking at ecosystem states, the type of fishing correlated positively with fishing impact, so ecosystems with both industrial and artisanal fisheries currently exhibit greater impacts of fishing than those dominated by industrial fisheries alone. Overall, our results concur with the temporal and spatial development of fishing activities from a concentration of effort in the North Atlantic, North Europe, and East Asia to a more globally distributed pattern with greater impacts on emergent fishing grounds located in Asia and Africa (Coll *et al*., 2008c; Libralato *et al*., 2008).

Caveats, limitations, and advantages

This study has presented a relatively simple analysis combining ecological indicators calculated from both states and trends, complemented with limited environmental and socio-economic input. The analyses provided notable results when they are compared with those of other studies analysing broader data or using ecosystem modelling as tools (e.g. Coll *et al*., 2006a, 2008b, c; Libralato *et al*., 2008; Shannon *et al*., 2009a, b).

Is it possible then to rank ecosystems using simple indicators? This question is not trivial because ecosystem complexity and differences in diversity mean that ecosystems can be very different in terms of their structure and dynamics. Ranking ecosystems along a unique axis from the least to the most impacted by fishing leads to integration of available data and facilitates communication with the general public, so might be an attractive objective. However, it is also a simplification that poses difficulties with interpretation. Although our indicators were selected primarily to capture the impact of fishing, they inevitably also reflect changes in the environment. Therefore, the final drivers for change in these indicators may be shared between fishing and the climate.

Nevertheless, changes in indicators generally suggest changes in the ecosystems towards overall situations of directions of a greater impact. To avoid excessive simplification, our study took into account not only the ecological indicators, but also the ecological and fisheries history of the ecosystems when interpreting the results, as well as drawing on available knowledge of ecosystem experts. In particular, the potential effects of other drivers, such as the environment, needed to be considered carefully when interpreting our results, especially when looking at upwelling or related systems (Shannon *et al*., 2010). Moreover, it is very likely that severely fished ecosystems may be more susceptible to environmental variability than more lightly fished ones, in accord with the notion that stressed ecosystems are less resilient to perturbation (Odum, 1985). The same has been noted for the Benguela ecosystems off southern Africa (Watermeyer *et al*., 2008a, b).

Further improvements to the ranking may be (i) to complement the list of the indicators to add environmental indicators and others capturing different human pressures, (ii) to increase the list of the ecosystem attributes to be taken into account, and (iii) to extend the list of the ecosystems being analysed.

A limitation of this work is the fact that all the ecosystems included for comparison were exploited. Therefore, our comparison is relative, and ecosystems cannot be classified between lightly and heavily impacted by fishing, but rather between those relatively lightly and those more heavily impacted by fishing. This conclusion is supported by Bundy *et al*. (2010), who found that all 19 ecosystems to be considered were impacted at the start of their time-series. A way forward would be to include ecosystems that have recovered either through a limited history of exploitation or good management practice. We could also calculate indicators for past ecosystem states using model-based descriptions (e.g. unfished states of the Benguela ecosystems; Watermeyer *et al*., 2008a, b) and compare these with model-based descriptions of present ecosystem states and data-based state indicators.

Some ecosystems were also limited in terms of the data available to calculate the ecological indicators. For example, indicators for the species-poor central Baltic Sea were calculated based only on the three main commercial fish (cod, herring, and sprat), which together constitute ∼90% of the catch and biomass. This is because of a lack of appropriate data, even for other commercial fish in the area (e.g. flatfish). The situation was the same for the southern Humboldt, where the indicators were calculated based only on four finfish species (horse mackerel, Araucanian herring, anchovy, and Chilean hake), which yield the bulk of the landings and are the main species for which continuous fishery-dependent and fishery-independent assessments are conducted in central Chile. Therefore, assessment of the central Baltic Sea and southern Humboldt within this manuscript should be considered as having some limitations that other ecosystems do not. However, the evaluation is valid for the main commercial fish and, because those play the key roles in the foodweb within these ecosystems (in term of energy and organic matter transfer), the assessment is probably still indicative of the broader ecosystem.

Other anthropogenic impacts of marine ecosystems, such as heavy eutrophication and pollution, were not considered. Such factors may be important in semi-enclosed ecosystems such as the central Baltic Sea and the north-central Adriatic Sea and need to be taken into account in future developments of comparative work such as that documented here.

It is necessary to be cautious in finally interpreting ecosystem rankings because of uncertainties related to the indicators. Blanchard *et al*. (2010) identified some sampling uncertainties linked to the data-collection processes as one of the potential factors making it difficult to detect significant trends in ecosystem indicators over short temporal scales. The link between the indicators and the practical conditions for their quantitative estimate must always be kept in mind in interpreting ecosystem indicator results (Jouffre *et al*., 2010).

It has been suggested that trends in indicators need to be compared against (logically based) reference directions as a means of codifying performance measures of an EAF. Our results showed that, although expected or reference trends should be borne in mind when analysing ecosystems in the context of an EAF, trends can change as a consequence of specific conditions of ecosystems and fisheries. This is certainly the case when lower-trophic-level organisms, such as small pelagic fish or invertebrates, are being heavily exploited or being strongly influenced by climate change. Some common and well understood indicators, such as fish size, lifespan, trophic level of the catch, or percentage of predatory fish, can highlight the fishing impacts. However, they may be interpreted erroneously as suggestive of recovery of ecosystems or high fishing impacts when there is directed fishing on lower-trophic-level organisms, or environmental perturbation of these. Close collaboration with local expertise from each system is therefore crucial to understanding and correctly interpreting ecosystem dynamics. Also, the use of a combination of target- and non-target-species indicators to analyse the current ecological state, recent trends, and long-term change is also essential.

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Appendix

Definition of the indicators used to rank marine ecosystems (based on Shin *et al.*, 2010a).

Indicator	Calculation and units
Mean length of fish in the community	$\bar{L} = \sum_i L_i/N$ (cm)
Trophic level of landed catch	$\overline{\mathrm{TL}}_{\mathrm{land}} = \sum_s \mathrm{TL}_s\, Y_s/Y$
Proportion of underexploited and moderately exploited stocks	Number of under- and moderately exploited stocks/total number of stocks considered
Proportion of predatory fish	Proportion of predatory fish = *B* predatory fish/*B* surveyed *B* surveyed = *B*(demersal fish + pelagic fish + commercially important invertebrates)
Mean lifespan	$\sum_S(\mathrm{age}_{\max} B_S)/\sum_S B_S$ (years)
1/*CV* of total biomass	Mean (total *B* for the past 10 years)/s.d. (total *B* for the past 10 years)
Total biomass of surveyed species	*B* (t)
Biomass/landings	*B*/*Y* retained species

L, length (cm); *i*, individual; *s*, species; *N*, abundance; *B*, biomass; *Y*, catch; *TL*, trophic level.

Surveyed species. Species sampled by researchers during routine surveys (as opposed to species sampled in catches by fishing vessels), including species of demersal and pelagic fish (bony and cartilaginous, small and large), and 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 are excluded. Surveyed species are those that are considered by default in the calculation of all survey-based indicators.

Retained species (landed). 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 certain size classes of that species may sometimes be discarded. A non-retained species is considered to be one that would never be retained for consumption 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.

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

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