

Ecological indicators to capture the effects of fishing on biodiversity and conservation status of marine ecosystems



M. Coll^{a,b,*}, L.J. Shannon^c, K.M. Kleisner^d, M.J. Juan-Jordá^{e,f}, A. Bundy^g, A.G. Akoglu^h, D. Banaruⁱ, J.L. Boldt^j, M.F. Borges^k, A. Cook^g, I. Diallo^l, C. Fu^j, C. Fox^m, D. Gascuelⁿ, L.J. Gurney^o, T. Hattab^p, J.J. Heymans^m, D. Jouffre^{q,r}, B.R. Knight^s, S. Kucukavsar^h, S.I. Large^d, C. Lynam^t, A. Machias^u, K.N. Marshall^v, H. Masski^w, H. Ojaveer^x, C. Pirotti^{b,y}, J. Tam^z, D. Thiao^A, M. Thiaw^B, M.A. Torres^{C,D}, M. Travers-Trolet^E, K. Tsagarakis^u, I. Tuck^{F,G}, G.I. van der Meer^H, D. Yemane^{c,I}, S.G. Zador^J, Y.-J. Shin^{a,c}

^a Institut de Recherche pour le Développement, CRH, Research Unit MARBEC (UMR 248), avenue Jean Monnet, CS 30171, 34203 Sète Cedex, France

^b Institute of Marine Science (ICM-CSIC), passeig Marítim de la Barceloneta, n° 37-49, 08003 Barcelona, Spain

^c Marine Research Institute and Department of Biological Sciences, University of Cape Town, Private Bag X3, Rondebosch, Cape Town 7701, South Africa

^d Northeast Fisheries Science Center, National Marine Fisheries Service, NOAA, 166 Water Street, Woods Hole, MA 02543, USA

^e AZTI, Marine Research Division, Herrera Kaia, Portualdea z/g, 20110 Pasaia, Gipuzkoa, Spain

^f Earth to Ocean Research Group, Department of Biological Sciences, Simon Fraser University, 8888 University Drive, Burnaby, BC V5A 1S6, Canada

^g Fisheries and Oceans Canada, Bedford Institute of Oceanography, 1 Challenger Drive, Dartmouth, NS B2Y 4A2, Canada

^h Middle East Technical University, Institute of Marine Sciences, P.O. Box 28, 33731 Erdemli, Mersin, Turkey

ⁱ Aix-Marseille Université, Mediterranean Institute of Oceanography (M.I.O.), UMR 7294, UR 235, Campus de Luminy, Case 901, 13288 Marseille Cedex 09, France

^j Fisheries and Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Road, Nanaimo, BC V9T 6N7, Canada

^k Instituto Português do Mar e da Atmosfera, Av. de Brasília, 1449-006 Lisboa, Portugal

^l Centre National des Sciences Halieutiques de Boussouira (CNSHB), 814 Rue MA500, Corniche sud Boussouira, BP 3738, Conakry, Guinea

^m Scottish Association for Marine Science, Scottish Marine Institute, Oban, Argyll PA37 1QA, UK

ⁿ Université Européenne de Bretagne, Agrocampus Ouest, UMR985 Ecologie et santé des écosystèmes, 65 route de Saint Brieuc, CS 84215, 35042 Rennes Cedex, France

^o Department of Earth, Ocean and Atmospheric Sciences, 2207 Main Mall, University of British Columbia, V6T 1Z4 BC, Canada

^p Jules Verne University of Picardie, Research Unit EDYSAN, FRE 3498 CNRS, 1 rue des Louvels, F-80037 Amiens Cedex 1, France

^q Institut de Recherche pour le Développement, Research Unit MARBEC, BP 1386, Dakar, Senegal

^r Laboratoire de Biologie et d'Ecologie des Poissons en Afrique de l'Ouest (LABEP-AO, IRD/IFAN), Institut Fondamental d'Afrique Noire, Campus universitaire UCAD, B.P. 206, Dakar, Senegal

^s Cawthron Institute, 98 Halifax Street East, Nelson 7010, New Zealand

^t Centre for Environment, Fisheries and Aquaculture Science, Pakefield Road, Lowestoft, Suffolk NR33 0HT, UK

^u Hellenic Centre for Marine Research, Institute of Marine Biological Resources and Inland Waters, Agios Kosmas, 16610 Elliniko, Athens, Greece

^v School of Aquatic and Fishery Sciences, University of Washington, Box 355020, Seattle, WA 98195, USA

^w Institut National de Recherche halieutique, Bd Sidi Abderrahmane, Casablanca, Morocco

^x Estonian Marine Institute, University of Tartu, Lootsi 2a, 80012 Pärnu, Estonia

^y Water Resources Unit, Institute for Environment and Sustainability, Joint Research Centre, Via E. Fermi 2749, Ispra (VA) 21027, Italy

^z Instituto del Mar del Perú (IMARPE), Esquina Gamarra y Gral. Valle s/n, Apartado 22, Callao, Lima, Peru

^A ISRA/Centre de Recherches Océanographiques de Dakar-Thiaroye (CRODT), BP 2241, Dakar, Senegal

^B Laboratoire d'Ecologie Halieutique-Afrique de l'Ouest (LEH-AO), ISRA/CRODT, Pôle de Recherche de Hann, BP 2241, Dakar, Senegal

^C Instituto Español de Oceanografía (IEO), Centro Oceanográfico de Cádiz, Puerto Pesquero, Muelle de Levante, s/n, P.O. Box 2609, E-11006 Cádiz, Spain

^D Swedish University of Agricultural Sciences, Department of Aquatic Resources, Institute of Coastal Research, Skolgtan 6, SE-742 42 Öregrund, Sweden

^E IFREMER, Fisheries Laboratory, 150 quai Gambetta, BP 699, 62321 Boulogne sur mer, France

^F National Institute of Water and Atmospheric Research Limited, Auckland, New Zealand

^G Department of Statistics, University of Auckland, Private Bag 92019, Auckland 1149, New Zealand

^H Institute of Marine Research, The Hjort Centre for Marine Ecosystem Dynamics, PB 1870, Nordnes, NO-5817 Bergen, Norway

^I Fisheries Branch, Department of Agriculture, Forestry, and Fisheries, Private Bag X2, Rogge Bay, 8012 Cape Town, South Africa

^J Alaska Fisheries Science Center, National Marine Fisheries Service, NOAA, Seattle, WA 98115, USA

* Corresponding author at: Institut de Recherche pour le Développement, CRH, Research Unit MARBEC (UMR 248), avenue Jean Monnet, CS 30171, 34203 Sète Cedex, France. Tel.: +33 0499573216.

E-mail address: marta.coll@ird.fr (M. Coll).

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ABSTRACT

IndiSeas (“Indicators for the Seas”) is a collaborative international working group that was established in 2005 to evaluate the status of exploited marine ecosystems using a suite of indicators in a comparative framework. An initial shortlist of seven ecological indicators was selected to quantify the effects of fishing on the broader ecosystem using several criteria (i.e., ecological meaning, sensitivity to fishing, data availability, management objectives and public awareness). The suite comprised: (i) the inverse coefficient of variation of total biomass of surveyed species, (ii) mean fish length in the surveyed community, (iii) mean maximum life span of surveyed fish species, (iv) proportion of predatory fish in the surveyed community, (v) proportion of under and moderately exploited stocks, (vi) total biomass of surveyed species, and (vii) mean trophic level of the landed catch. In line with the Nagoya Strategic Plan of the Convention on Biological Diversity (2011–2020), we extended this suite to emphasize the broader biodiversity and conservation risks in exploited marine ecosystems. We selected a subset of indicators from a list of empirically based candidate biodiversity indicators initially established based on ecological significance to complement the original *IndiSeas* indicators. The additional selected indicators were: (viii) mean intrinsic vulnerability index of the fish landed catch, (ix) proportion of non-declining exploited species in the surveyed community, (x) catch-based marine trophic index, and (xi) mean trophic level of the surveyed community. Despite the lack of data in some ecosystems, we also selected (xii) mean trophic level of the modelled community, and (xiii) proportion of discards in the fishery as extra indicators. These additional indicators were examined, along with the initial set of *IndiSeas* ecological indicators, to evaluate whether adding new biodiversity indicators provided useful additional information to refine our understanding of the status evaluation of 29 exploited marine ecosystems. We used state and trend analyses, and we performed correlation, redundancy and multivariate tests. Existing developments in ecosystem-based fisheries management have largely focused on exploited species. Our study, using mostly fisheries independent survey-based indicators, highlights that biodiversity and conservation-based indicators are complementary to ecological indicators of fishing pressure. Thus, they should be used to provide additional information to evaluate the overall impact of fishing on exploited marine ecosystems.

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1. Introduction

Changes in marine resources and ecosystems have been documented worldwide (Butchart et al., 2010; Lotze et al., 2006) and multiple anthropogenic and climate-related drivers of change have been identified (Halpern et al., 2008). These drivers can alter ecosystem structure and functioning (Christensen et al., 2003; Frank et al., 2005) and can affect the ecosystem services that humans obtain from healthy oceans (Worm et al., 2006). Consequently there is growing concern about the status of marine ecosystems and a need to define, test and prioritize robust indicators to track ecosystem status to inform management decisions.

In the marine science research field, there has been considerable discussion about how to define, calculate, prioritize, test and evaluate indicators to monitor the pressures on, and status of exploited marine ecosystems (e.g., Rombouts et al., 2013; Shin et al., 2010a). Initially, indicators were developed to include ecological considerations with the goal of capturing the impact of dominant pressures, such as fishing or eutrophication (Cury et al., 2005; de Leiva Moreno et al., 2000). However, recently the scope of ecosystem indicators has expanded to include socio-economic and governance issues and the cumulative impacts of multiple human activities (e.g., Boldt et al., 2014; Halpern et al., 2012; Large et al., 2015; Levin et al., 2009; Tittensor et al., 2014).

Fishing represents one of the greatest pressures on marine ecosystems (Costello et al., 2010), and ecological indicators have been used to quantify its impacts on the status of ecosystems and to provide the rationale for scientific advice. Progress has included the establishment of criteria and frameworks to: (i) guide the selection of indicators (e.g., Rice and Rochet, 2005) that are used to assess the effects of fishing via trend (e.g., Blanchard et al., 2010; Coll et al., 2010b) and threshold (Large et al., 2013) analyses, (ii) define preliminary reference levels and reference directions for selected indicators (e.g., Link et al., 2002; Shin et al., 2010a), and (iii) develop and test evaluation frameworks (e.g., Bundy et al., 2010; Kleisner et al., 2013).

In 2005, the *IndiSeas* (“Indicators for the Seas”) Working Group was initiated under the auspices of the European Network of Excellence, Eur-Oceans. *IndiSeas* followed from the Scientific Committee on Oceanic Research of the Intergovernmental Oceanographic Commission (SCOR/IOC) Working Group on “Quantitative Ecosystem Indicators” (Shin and Shannon, 2010; Shin et al., 2010b, www.indiseas.org). During the first phase of *IndiSeas* (2005–2010, hereafter *IndiSeas*-phase I), the goals were to perform analyses of ecological indicators to quantify the impact of fishing on the status of exploited marine ecosystems in a comparative framework and to provide decision support criteria for an Ecosystem Approach to Fisheries (EAF) by means of a common suite of interpretation and visualization methods. The rationale was that, although the current primary objective of fisheries management is to ensure sustainable levels of harvest for commercial stocks, the incorporation of broader ecosystem considerations into managing fisheries has become an increasingly important obligation in many countries and regions throughout the world (e.g., Link, 2002; Murawski, 2000; Pikitch et al., 2004; Walters et al., 2005).

Thus, in *IndiSeas*-phase I, a suite of empirical ecological indicators was selected using several criteria (ecological meaning, sensitivity to fishing, data availability, management objectives and public awareness), to create a shortlist of indicators that were easy to calculate from landings and surveys data and that were meaningful and comparable across many marine ecosystems worldwide (Shin et al., 2012). These indicators were: (i) the inverse coefficient of variation of total biomass in the surveyed community (also referred to “Biomass Stability”, or BS), (ii) mean fish length in the surveyed community (“Fish Size”, LG), (iii) mean maximum life span of surveyed fish species (“Life Span”, LS), (iv) proportion of predatory fish in the surveyed community (“Predators”, PF), (v) proportion of under and moderately exploited stocks (“Sustainable Stocks”, SS), (vi) total biomass of surveyed species (“Biomass”, TB), and (vii) mean trophic level of the landed catch (“Trophic Level”, TLc) (Table 1). All the indicators are survey-based with the exception of SS and TLc. In previous studies these indicators were

Table 1

IndiSeas-phase I ecological indicators used to track the impacts of fishing on exploited marine ecosystems and *IndiSeas*-phase II new ecological indicators used to track the broader impacts of fishing on exploited marine ecosystems in relation to biodiversity and conservation-based issues (see Table S1 for details).

<i>IndiSeas</i> indicators		Label	Acronym	Used for state or trend	Survey (S), catch (C) or model based (M)
Phase I					
1	1/coefficient of variation of total biomass of surveyed species	Biomass stability	BS	S	S
2	Mean fish length in the surveyed community	Fish size	LG	S, T	S
3	Mean maximum life span of surveyed fish species	Life span	LS	S, T	S
4	Proportion of predatory fish in the surveyed community	Predators	PF	S, T	S
5	Proportion of under and moderately exploited stocks	Sustainable stocks	SS	S	C
6	Total biomass of surveyed species	Biomass	TB	T	S
7	Mean trophic level of the landed catch	Trophic level	TLc	S, T	C
Phase II					
1	Mean intrinsic vulnerability index of the fish landed catch	Mean vulnerability	IVI	T	C
2	Proportion of non-declining exploited species	Non-declining species	NDES	S	S
3	Catch-based marine trophic index	Trophic index	MTI	S, T	C
4	Mean trophic level of the surveyed community	Trophic level of the community	TLsc	S, T	S
5	Mean trophic level of the modelled community	Trophic level of the model	TLmc	S, T	M
6	Proportion of discards in the fishery	Landings/Discards	D	S, T	C

calculated for 19 exploited marine ecosystems, which included temperate, tropical, upwelling, and high latitude ecosystems. Comparative analyses of these indicators provided insights on the relative states and trends of these ecosystems given fishing pressures exerted upon them (e.g., Blanchard et al., 2010; Bundy et al., 2010; Coll et al., 2010b; Link et al., 2010; Shin et al., 2010a).

These comparative studies elucidated the need to expand the list of *IndiSeas*-phase I indicators to cover additional dimensions of the impacts of fishing, such as socioeconomic and governance interactions, to include the effects of a variable and changing environment, and to emphasize the broader biodiversity and conservation risks of fishing when evaluating the status of marine ecosystems (Bundy et al., 2012; Shin et al., 2012). Socioeconomic and environmental factors are addressed in the second phase of *IndiSeas* (2010–2014, hereafter *IndiSeas*-phase II), endorsed by IOC/UNESCO. Here we focus on the scientific challenges posed by the Nagoya Strategic Plan of the Convention on Biological Diversity (2011–2020) (CBD, 2010) by emphasizing and testing the utility of key biodiversity and conservation-based indicators while accounting for trade-offs between different societal goals (e.g. conservation of biodiversity; sustainable exploitation) incurred in the management of marine ecosystems (Brander, 2010; Palumbi et al., 2008). Some of these biodiversity and conservation-based indicators can help illustrate important conservation implications and can contribute to the evaluation of progress towards achieving the biodiversity-related “Aichi Targets” (Tittensor et al., 2014).

Here we first present the additional suite of biodiversity and conservation-based indicators studied in *IndiSeas*-phase II and the rationale underlying their inclusion. Next, we examine the whole suite of indicators across 29 exploited marine ecosystems distributed worldwide and assess whether any of the indicators are correlated and potentially redundant. We then use a comparative approach to evaluate the status of these ecosystems using the whole suite of indicators. Considering the complexity of marine ecosystems, the scale and scope of change manifested and the difficulty of undertaking controlled experiments, comparative analysis of ecosystems is expected to provide insight on how drivers influences dynamics of ecosystems (Murawski et al., 2009). In our case, this allows us to assess whether the additional biodiversity and conservation-based indicators provide new insights on the status of exploited marine ecosystems. Finally, we test whether fishing pressure is correlated with changes observed in our suite of ecological indicators by investigating the relationship between indicator trends and three measures of fishing pressure.

Our overall objective is to present a comprehensive suite of ecological indicators with the greatest potential to capture

broad biodiversity and conservation considerations of fishing on exploited marine ecosystems. Based on the examination of the suite of ecological indicators for several ecosystems, we discuss the best subset of indicators that would complement the previously selected ecological indicators of *IndiSeas*-phase I. In addition, we contribute to the evaluation of the status of exploited ecosystems, which is necessary for balancing conservation and fishing objectives in marine ecosystems.

2. Material and methods

2.1. Case studies

Our analyses used 29 exploited marine ecosystems as case studies (Fig. 1 and Table 2). They correspond to upwelling, high latitude, temperate and tropical marine ecosystems, and cover a range of low to high productivity areas, located in the Atlantic and Pacific Oceans, and the Mediterranean, Black and Baltic Seas. A key strength of the *IndiSeas* approach lies in the participation of ecosystem experts who provide local data and specific, local interpretation of the indicators and who can inform comparisons and analyses of any biases in data collection or generation of indicator results (Shin et al., 2010b, 2012). This study takes full advantage of these expertise and ecosystem experts provided insights to interpret indicator scores.

2.2. Selection of biodiversity and conservation-based indicators

We used a step-by-step process to select indicators, as done in *IndiSeas*-phase I (Shin and Shannon, 2010; Shin et al., 2010b), to augment the original indicators suite with additional biodiversity and conservation-based metrics that would capture the broader effects of fishing on marine biodiversity and ecosystem functioning (Table 1 and Table S1). The selection process included the following steps: (i) potential indicators were identified by reviewing the scientific literature, (ii) indicators were evaluated with the screening criteria, and (iii) a suite of potential biodiversity-and conservation-based ecological indicators was proposed for examination in a subset of comparable ecosystem case studies. First, a list of potential indicators was identified from the scientific literature for consideration with no restriction on the number of indicators. These indicators were subjected to screening criteria by experts so that each candidate indicator was scored by local experts for 20 different ecosystems, and scores were averaged per criteria for each indicator (Table S2). Screening criteria comprised data availability,

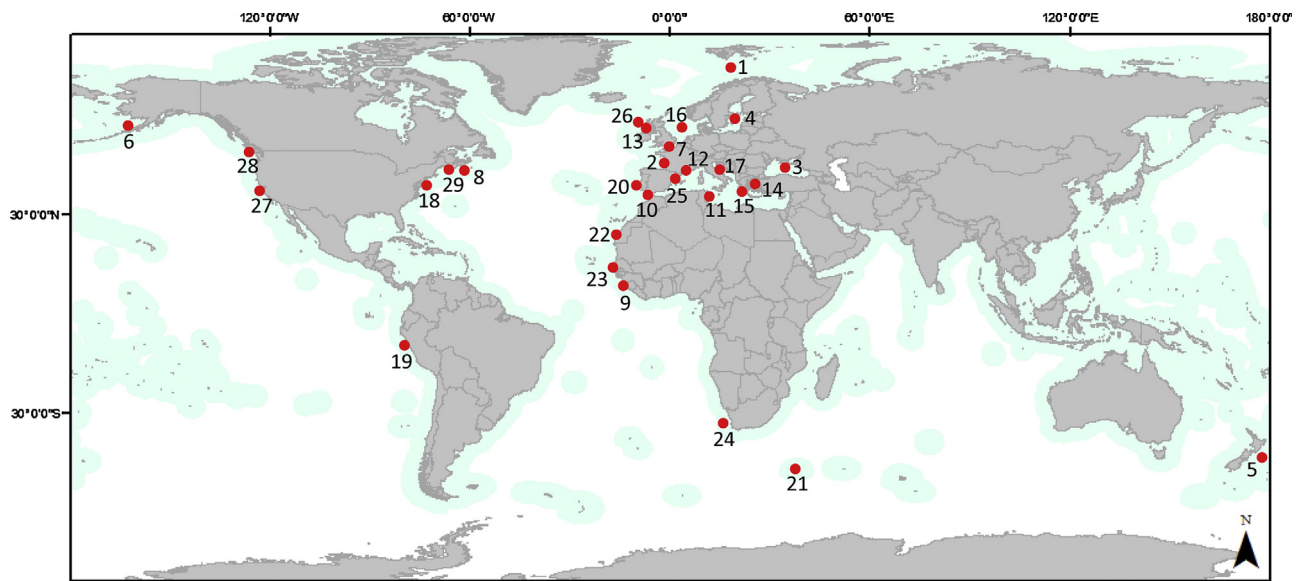


Fig. 1. Location of the 29 case studies of exploited marine ecosystems included in the analyses (ecosystem names are listed in Table 2).

measurability, ecological meaning, sensitivity to fishing, management objectives, and public awareness (Shin and Shannon, 2010).

Supplementary Tables S1 and S2 related to this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ecolind.2015.08.048>.

As a result of this process, the additional biodiversity and conservation-based indicators chosen to supplement the initial *IndiSeas*-phase I indicators (Shin et al., 2010a,b) were: (i) mean intrinsic vulnerability index of fish in the landed catch ("Mean Vulnerability", or IVI) (Cheung et al., 2007), (ii) proportion of non-declining surveyed exploited species ("Non-Declining Exploited Species", NDES) (Kleisner et al., 2015), (iii) catch-based marine trophic index ("Trophic Index", MTI) (Pauly and Watson, 2005), and (iv) mean trophic level of the surveyed community

("Trophic Level of the Community", TLsc) (Shannon et al., 2014) (Table 1 and Table S1). In addition, two extra indicators were chosen (for ecosystems with sufficient data): (v) mean trophic level of the modelled community ("Trophic Level of the Model", TLmc, calculated using Ecopath with Ecosim food web models) (Shannon et al., 2014); and (vi) proportion of discards in the fishery ("Landings/Discard", D) (Fulton et al., 2005; Link, 2005; Shannon et al., 2014). Hereafter, we referred to this new proposed suite of six additional biodiversity-and conservation-based ecological indicators as *IndiSeas*-phase II indicators. The Non-Declining Exploited Species indicator was recently explored in a subset (22) of *IndiSeas* ecosystems included in the present analysis (Kleisner et al., 2015) so we build upon results of that study.

Table 2

List of 29 exploited marine ecosystems used in the analyses (Fig. 1).

	Ecosystems	Label	Geographical area	Ecosystem type
1	Barents Sea	BarentsS	NE Atlantic Ocean	High latitude
2	Biscay Bay	BiscayB	NE Atlantic Ocean	Temperate
3	Black Sea	BlackS	Black Sea	Temperate
4	Central Baltic Sea	CBalticS	NE Atlantic Ocean	Temperate
5	Chatham Rise	ChathamR	SW Pacific Ocean	Temperate
6	Eastern Bering Sea	EBeringS	NE Pacific Ocean	High latitude
7	Eastern English Channel	EEnglishC	NE Atlantic Ocean	Temperate
8	Eastern Scotian Shelf	EScotianS	NW Atlantic Ocean	Temperate
9	Guinean Shelf	GuineaS	E Central Atlantic Ocean	Tropical
10	Gulf of Cadiz	GoC	NW Atlantic Ocean	Temperate
11	Gulf of Gabes	GoG	C Mediterranean Sea	Temperate
12	Gulf of Lions	GoL	NW Mediterranean Sea	Temperate
13	Irish Sea	IrishS	NE Atlantic Ocean	Temperate
14	North Aegean Sea	NAegeanS	E Mediterranean Sea	Temperate
15	North Ionian Sea	NlonianS	C Mediterranean Sea	Temperate
16	North Sea	NorthS	NE Atlantic Ocean	Temperate
17	North-central Adriatic Sea	NCA AdriaticS	C Mediterranean Sea	Temperate
18	North-east U.S.	NEUS	NW Atlantic Ocean	Temperate
19	Northern Humboldt Current	NHumboldtC	SE Pacific Ocean	Upwelling
20	Portuguese Coast	PortugalC	NE Atlantic Ocean	Temperate
21	Prince Edward Islands	PEI	S Indian Ocean	High latitude
22	Sahara Coastal	SaharaC	E Central Atlantic Ocean	Upwelling
23	Senegalese Shelf	SenegalS	E Central Atlantic Ocean	Upwelling
24	Southern Benguela	SBenguela	SE Atlantic Ocean	Upwelling
25	Southern Catalan Sea	SCatalanS	NW Mediterranean Sea	Temperate
26	West Coast Scotland	WCSotland	NE Atlantic Ocean	Temperate
27	West Coast U.S.	WCUS	NE Pacific Ocean	Upwelling
28	West Coast Vancouver Island	WCVancouverI	NE Pacific Ocean	Upwelling
29	West Scotian Shelf	WScotianS	NW Atlantic Ocean	Temperate

All indicators were formulated so that a decrease in their value is expected with greater fishing pressure. Thus, the lowest value of the indicator, or a decrease of the indicator with time, would theoretically indicate a higher impact of fishing on the ecosystem. Indicators were used to represent the current state of the ecosystem and/or trend over time (Table 1).

2.3. Analyses of indicators

Indicators were calculated for the 29 exploited marine ecosystems included in this analysis (Table 2 and Fig. 1) using landings and survey data provided by local experts. Using the whole suite of indicators, we derived common metrics: (i) the current state of the indicators, and (ii) the overall trends of the indicators. These common metrics were used to evaluate whether any of the indicators were correlated and potentially redundant, and to conduct a comparative study across marine ecosystems.

2.3.1. Analyses of current states and overall trends

We calculated the current state indicators as the mean of the most recent five years for which data were available (for most systems this was 2005–2010) to provide a measure of the current state of the ecosystem. State indicator patterns were visualized using heat maps and petal plots, where values were standardized between 0 and 1, based on the minimum and maximum values found across all ecosystems.

We examined trends in indicators for years during 1980–2010, or for the years within this period for which data were available (Figure S7). We used two methods to quantify the overall direction of change for each indicator. The first method assumed linearity over time, using a generalized linear model and accounting for autocorrelation, where present, to fit a trend. The second method allowed for the possibility of non-linearity over time and measured the overall trend based on the average rate of change across all years included (i.e., rate of increase or decrease between multiple consecutive years). Since indicator series differed in time coverage and time span due to data availability, only indicator series having at least two consecutive years within a time series of data were used in this analysis. Trend indicators were visualized using heat maps of slopes and average rates of change if the trends over time and their significance where values were scaled between 0 and 1, based on the minimum and maximum values found across all ecosystems.

All state and trend analyses were conducted in R version 3.0.2 (R Core Team 2013).

(i) *Analysis of trends assuming linearity over time:* We fit a generalized, least-squares regression model to each indicator time series, first testing and correcting for autocorrelation where present (following Blanchard et al., 2010; Coll et al., 2008). Trends were estimated using time series of normalized indicator values to allow comparison of trends (Blanchard et al., 2010), standardized by subtracting the mean and dividing by the standard deviation. This standardization allows the indicators to be expressed on the same scale and with the same spread.

A two-stage estimation procedure was used to take into account temporal autocorrelation in the residuals and to satisfy regression assumptions (Coll et al., 2008). This procedure was generally sufficient for trend estimation as the time-series were relatively short and there was considerable flexibility in realizations of the autocorrelated errors. We assessed the significance of the estimated trend (p -value), the direction of the trend (positive or negative slope) and the magnitude of the slope.

(ii) *Analysis of trends allowing for non-linearity over time:* To allow for the possibility of non-linearity over time in the indicators, we used a two-step estimation procedure to calculate the average annual rate of change for each indicator across all the years. First,

we converted the raw time series of each indicator to successive annual rates of change (r_i) (Juan-Jordá et al., 2011):

$$r_i = \ln \cdot \left(\frac{I_{i+1}}{I_i} \right) \quad (1)$$

where I_i is the indicator value in time i and I_{i+1} is the value of the indicator a year later ($i + 1$).

This method of estimating the ratios in log-scale enables the indicators to be expressed on the same scale, thus rendering them unitless. This is a common means of removing temporal autocorrelation from a time series (Shumway and Stoffer, 2006). Therefore, unlike the first method, the indicators were not standardized for spread, but have equivalent units.

We then estimated the average of the annual rates of change across all the successive years for each indicator to obtain a metric of the overall rate of change of each indicator using the following model form:

$$r_i = \beta_0 + e_i \quad (2)$$

where r_i , the dependent variable, is the successive annual (i) rate of change between two consecutive years in each indicator; β_0 , the model intercept, is the model average annual rate of change in each indicator across all the years, and e_i is the normally distributed residual error. We assessed the significance of β_0 , the model average annual rate of change across all the years (p -value), the direction of the rate of change (positive or negative) and the magnitude of the rate of change.

2.3.2. Complementarity and redundancy analyses

We performed separate analyses to test for correlation across state and trend indicators among all ecosystems in order to identify complementarity and redundancy in the indicators selected. All correlations were evaluated using the Spearman's non-parametric rank order correlation coefficient, which is a measure of statistical dependence between two variables, ranging between -1 and 1 , i.e., perfect negative and positive correlation, respectively. This test assesses how well the relationship between two variables can be described using a non-linear monotonic function. Moreover, correlation coefficients among trends were summarized as a matrix of positive or negative correlations between indicators for all ecosystems to quantify the proportion of trends with a significant change and assess the overall redundancy. These correlation analyses allowed us to evaluate the suitability of our suite of indicators to track the different ecosystem effects of fishing and whether we need to retain the full suite for further analyses. These analyses were performed using R version 3.0.2 (R Core Team 2013).

2.3.3. Comparative approach to diagnose the exploitation status of marine ecosystems

The current state and the magnitude, direction, and significance of the trends of each indicator were used to compare the 29 case study ecosystems following a similar methodology to that in a previous comparative analysis, which ranked ecosystems in terms of their exploitation level (Coll et al., 2010b; Shannon et al., 2009).

We first used the heat maps and petal plots to compare the current state of each indicator across all the ecosystems. We then used heat maps to compare trends, including magnitude, direction and significance of trends of each indicator across all the ecosystems. Subsequently, we used non-parametric multivariate analyses (cluster analysis and non-metric MultiDimensional Scaling, nMDS) to perform a synthetic comparison of all ecosystems based on their similarity. These analyses were performed using *IndiSeas*-phase I indicators and then the whole suite of indicators so additional information on ecosystem status from *IndiSeas* II indicators could be assessed. We evaluated the suitability of the suite of indicators and whether it was necessary to retain the full suite for further

analyses. All multivariate analyses were performed with PRIMER v6 (Clarke and Gorley, 2006). Because the indicators have different units and scales, we normalized the data prior to the construction of the Euclidean distance matrices (Clarke and Gorley, 2006).

2.3.4. Correlation analyses with fishing pressure

Using Spearman's non-parametric rank correlations, we cross-correlated time series of fishing pressure indicators and our suite of ecological indicators used for trend analyses. First, we investigated the relationship between the trends in the suite of ecological indicators and a global fishing pressure indicator (the ratio of landings to survey biomass, L/B). This indicator had been selected in *IndiSeas*-phase I as it was simple and most readily available pressure indicator across the ecosystems examined at that time (Shin et al., 2010b) (Figure S1). In *IndiSeas*-phase II, relative fishing effort and relative fishing mortality were also available for a subset of nine marine ecosystems (Shannon et al., 2014, Figure S2). Therefore, we used a non-weighted mean of the relative fishing effort across fleets and species and a non-weighted mean of the fishing mortality rate across species in order to test the correlations between our suite of pressure indicators of fishing pressure and our suite of ecological indicators. All correlations were evaluated using Spearman's non-parametric rank order correlation coefficient in R version 3.0.2 (R Core Team 2013).

Supplementary Figures S1 and S2 related to this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ecolind.2015.08.048>.

3. Results

3.1. State indicators

The current state (2005–2010) of *IndiSeas*-phase I indicators across all the ecosystems varied greatly (Fig. 2 and Figures S3 and S4). The scores of most of the indicators were relatively low (more indicators showing values <0.5). For 19 ecosystems (66% of the ecosystems): the Bay of Biscay, the central Baltic Sea, the eastern English Channel, the Guinean Shelf, the Gulf of Cadiz, the Gulf of Gabes, the Gulf of Lions, the Irish Sea, the north Aegean Sea, the north Ionian Sea, the North Sea, the north-central Adriatic Sea, the northern Humboldt Current, the Portuguese Coast, the Sahara Coastal, the Senegalese Shelf, the southern Benguela, the southern Catalan Sea, the western Scotian Shelf, suggesting a more impacted ecosystem state on average compared to other ecosystems. In two ecosystems (7%), the scores for most of the indicators were relatively high (more indicators showed values higher than 0.5): the eastern Bering Sea and the west Coast Vancouver Island, suggesting these ecosystems have a less impacted ecosystem state overall. For 7 ecosystems (24%), the current state of the indicators varied, producing a mixed signal: the Barents Sea, the Chatham Rise, the eastern Scotian Shelf, the northeast U.S., the Prince Edward Islands, the west Coast Scotland and the western Coast U.S. The Black Sea had data available for only one indicator (TLc) in the recent years. The Prince Edward Islands lacked data for four of the six state indicators and nine ecosystems were missing data for the Fish Size (LG) indicator (Fig. 2).

Supplementary Figures S3 and S4 related to this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ecolind.2015.08.048>.

The current state (2005–2010) in the *IndiSeas*-phase II indicators across all the ecosystems also varied greatly (Fig. 2 and Figures S5 and S6). In eight ecosystems (28%) the scores of most of the indicators were relatively low (<0.5) suggesting a more impacted ecosystem state on average compared to the other ecosystems: the Black Sea, the Gulf of Cadiz, the north Aegean Sea, the north Ionian

Sea, the north-central Adriatic Sea, the northern Humboldt Current, the Senegalese shelf and the southern Catalan Sea. In 13 ecosystems (45%) the scores for most of the indicators were relatively high (>0.5), suggesting these ecosystems have a less impacted ecosystem state: the Barents Sea, the Bay of Biscay, the Chatham Rise, the eastern Bering Sea, the eastern English Channel, the eastern Scotian Shelf, the Gulf of Lions, the North Sea, the northeast U.S., the Portuguese Coast, the southern Benguela, the west Coast U.S. and the west Coast Vancouver Island. For six ecosystems (21%) the indicators showed contrasting patterns: the central Baltic Sea, the Guinean Shelf, the Gulf of Gabes, the Irish Sea, the west Coast Scotland, and the western Scotian Shelf. There was not enough data to assess the state in the Sahara Coastal and the Prince Edward Islands because they only had data for a single indicator. The two extra indicators, Landings/Discards and Trophic Level of the Model, were only available in nine and eleven ecosystems, respectively (Fig. 2), and showed a dominance of low values for those ecosystems with data available (thus higher impacts).

Supplementary Figures S5 and S6 related to this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ecolind.2015.08.048>.

The combined assessment of *IndiSeas*-phase I and II indicators produced similar results for 12 ecosystems (41%) (Fig. 2 and Figures S3 and S5). Indicators were comparatively low (<0.5) for both suites of indicators in nine ecosystems: the Black Sea, the Gulf of Cadiz, the north Aegean Sea, the north Ionian Sea, the north-central Adriatic Sea, the northern Humboldt Current, the Sahara Coastal, the Senegalese shelf and the southern Catalan Sea. Two ecosystems showed generally high indicators (>0.5) in suites of indicators: the eastern Bering Sea and the west Coast Vancouver Island, and one ecosystem showed mixed signals: west Coast Scotland. In 59% of the ecosystems examined, high values for phase I indicators did not always correspond to high values for phase II indicators. For example, some upwelling systems such as the southern Benguela had higher scores for on *IndiSeas*-phase II indicators compared to the *IndiSeas*-phase I indicators. Similar results were evident for Mediterranean systems such as the Gulf of Lions or the Gulf of Gabes.

3.2. Trend indicators

Between 1980 and 2010, the overall direction of change of *IndiSeas*-phase I indicators varied greatly among ecosystems (Fig. 3 and Figure S7). Six ecosystems (21%) showed an overall decrease in the levels of indicators, suggesting an overall increasingly impacted ecosystem over time: the Black Sea, the central Baltic Sea, the Guinean Shelf, the Sahara Coastal, the Gulf of Cadiz and the west Coast U.S. Three ecosystems (10%) showed an overall increase, suggesting these ecosystems have become less impacted over time: the Barents Sea, the Gulf of Lions and the west Coast Scotland. Ten ecosystems (35%) showed mixed signals, with some indicators increasing and others decreasing significantly: the eastern Scotian Shelf, the Irish Sea, the north Ionian Sea, the north-central Adriatic Sea, the northeast U.S., the northern Humboldt Current, the Portuguese Coast, the Senegalese Shelf, the southern Benguela, and the western Scotian Shelf. Indicator scores for ten ecosystems (35%) did not show any clear patterns because only one indicator showed a significant trend (increasing or decreasing). Results using *IndiSeas*-phase II indicators were similar to those of *IndiSeas* phase-I indicators (Fig. 3 and Figure S7). Five ecosystems (17%) showed an overall decrease: the central Baltic Sea, the eastern Scotian Shelf, the north-central Adriatic Sea, the Portuguese Coast and the Prince Edward Islands. Two ecosystems (7%) showed an overall increase: the Barents Sea and the north Ionian Sea. Eight ecosystems (28%) showed mixed signals because indicators either increased or decreased significantly: the Irish Sea, the north Aegean Sea, the northern Humboldt Current, the Senegalese Shelf, the southern Catalan Sea,

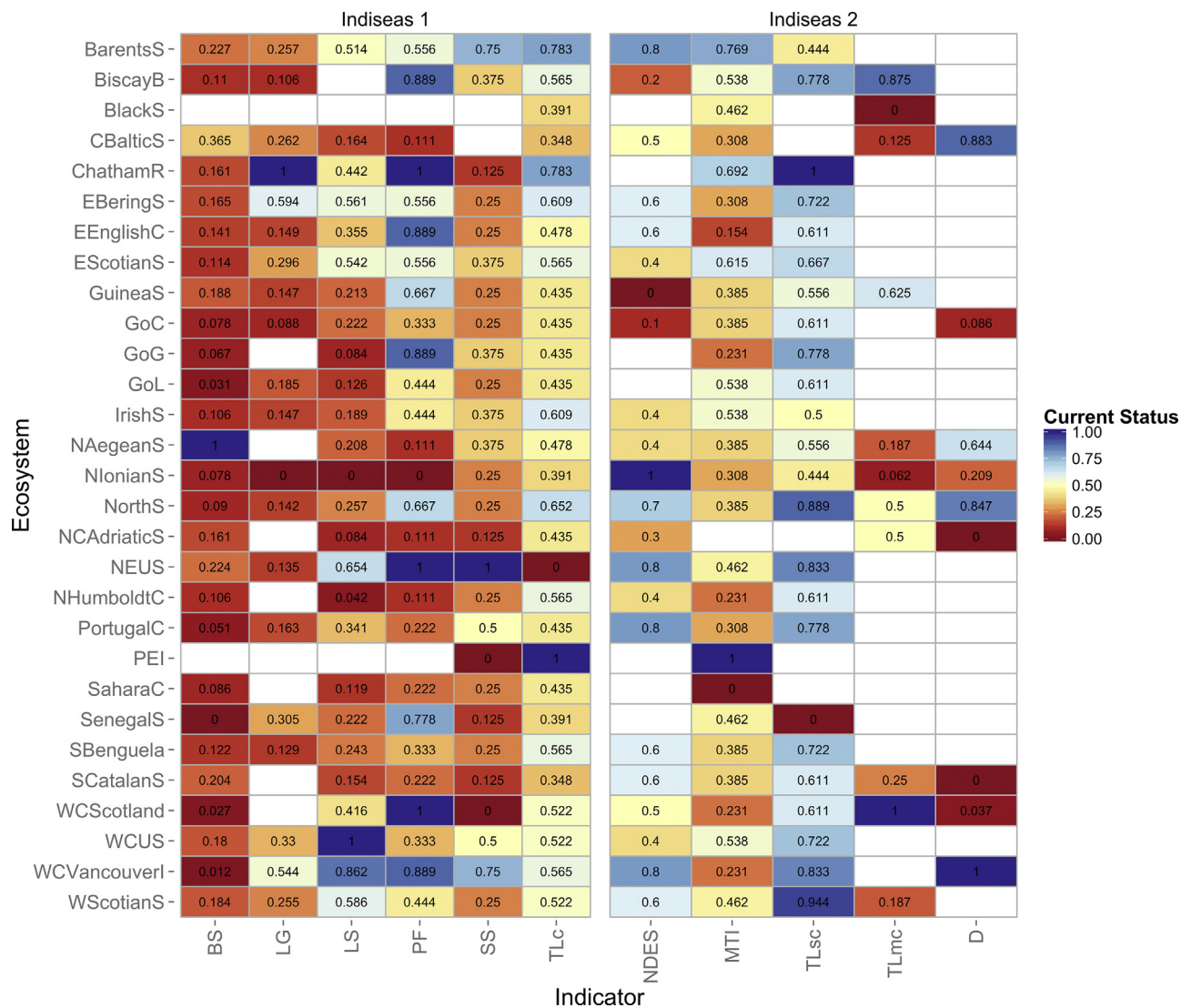


Fig. 2. Heatmap of current state indicators (2005–2010) using both *IndiSeas*-phase I (left panel) and II (right panel) indicators (Table 1). Indicator values are scaled between 0 and 1, based on the minimum (red) and maximum (blue) values found across all ecosystems. Full indicator names and acronyms are listed in Table 1 and ecosystem names and labels are listed in Table 2. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of the article.)

the west Coast Scotland, the west Coast U.S. and the western Scotian Shelf, while 14 ecosystems (49%) did not show any clear pattern due to the fact that only one indicator changed significantly.

Supplementary Figure S7 related to this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ecolind.2015.08.048>.

The joint comparison of trends in *IndiSeas*-phase I and phase II indicators illustrated that overall trends were similar between the two suites of indicators for 16 ecosystems (55%) (Fig. 3). One ecosystem, the Barents Sea, showed an increasing trend in the two suites of indicators, while one ecosystem, the central Baltic Sea, showed a consistent decreasing trend in both suites of indicators. In addition, four ecosystems showed consistent mixed signals: the Irish Sea, the northern Humboldt Current, the Senegalese Shelf and the western Scotian Shelf. Ten ecosystems showed no overall pattern of change in either one or the other suite of indicators: the eastern Bering Sea, the Bay of Biscay, the Black Sea, the Chatham Rise, the eastern English Channel, the Gulf of Gabes, the Gulf of Lions, the Sahara Coastal, the North Sea, the western Coast Vancouver Island. The other 13 ecosystems showed different trends when comparing *IndiSeas*-phase I with phase II indicators.

Most *IndiSeas*-phase I and phase II indicators across the majority of ecosystems showed a non-significant overall direction of change when comparing the rates of change over time (Fig. 4). This method is more sensitive to time series with low signal to noise ratio (indicators which are more variable over time) resulting in a lower detection of significant trends (Figure S7). However, because the indicators were not corrected for differences in spread with this method, the ecological significance of small changes in indicator values is unknown. Only one or two indicators showed a significant declining average annual rate of change over time in four ecosystems. In the central Baltic Sea, the Trophic Level of the catch had decreased on average -0.3% per year and the Mean Vulnerability had decreased on average -0.2% per year over the time period considered. In the southern Benguela, the Trophic Level of the Model had decreased on average -0.4% per year, and in the Guinean Shelf ecosystem this indicator had declined on average -0.1% per year. In the west Coast U.S., Biomass had declined on average -6.4% per year over the time period considered.

Although the sensitivity of the two methods used to estimate overall trends in indicators varied greatly in terms of detecting significance (Figs. 3 and 4), we found that in eight ecosystems (28%) all trends showed the same positive or negative directions and

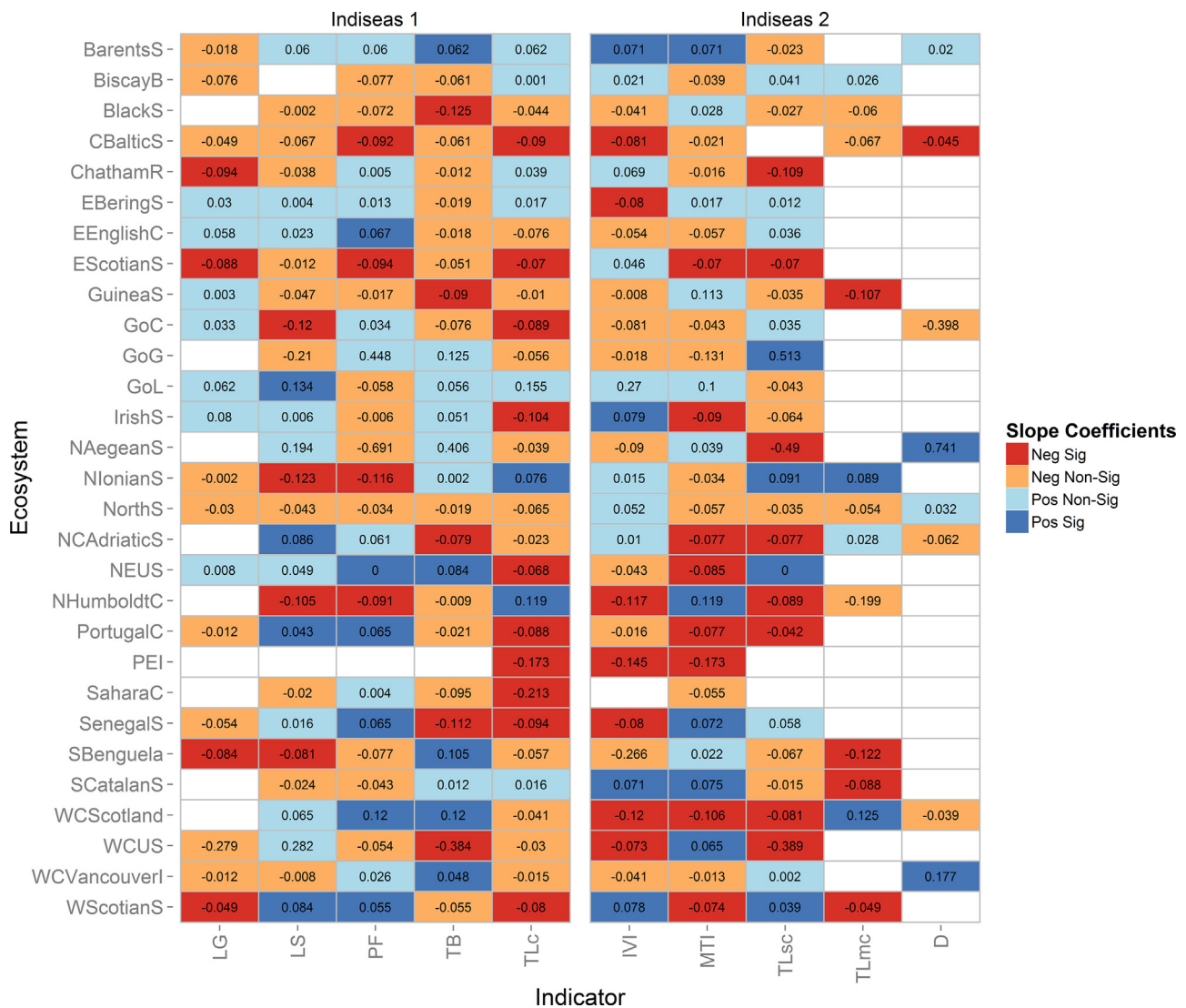


Fig. 3. Heatmap of trend indicators' slope coefficients (1980–2010, Figure S7) using both *IndiSeas*-phase I (left panel) and II (right panel) indicators (Table 1) and the generalized least-squares and autoregressive error analysis (assuming linearity over time). Neg: negative, Pos: positive, Sig: significant, Non-Sig: non-significant trend. Full indicator names and acronyms are listed in Table 1 and ecosystem names and labels are listed in Table 2.

in 13 ecosystems (45%) trends showed similar directions, differing in only one or two indicators per ecosystems. In several cases (e.g., the Southern Benguela), more negative (although often non-significant) trends were identified using the average rates of change method of trend detection.

3.3. Complementarity and redundancy of indicators

With respect to state indicators averaged over the five most recent years, positive and significant correlations between Life Span and Predators, Life Span and Sustainable Stocks, and Trophic Level of the catch and Fish Size from the *IndiSeas*-phase I state indicators highlighted some redundancy between indicators (Table 3). No significant correlations were found between state indicators of the second suite from *IndiSeas*-phase II. We observed three significant positive correlations between *IndiSeas*-phase I and phase II state indicators (Table 3): Predators and Trophic Level of the Model, Sustainable Stock and Landings/Discards, and Trophic Level of the catch and Trophic Index (MTI). No strong negative correlations were registered between indicators, which suggested that indicators did not show conflicting results in different ecosystems.

With respect to the trend indicators, more than half of the ecosystems present a positive and significant correlation between Life Span and Predators from the *IndiSeas*-phase I indicators, which highlighted some redundancy between these particular indicators (Table 4), as in the analysis of state indicators. Similarly, we observed low proportions (lower than 50%) of non-significant correlations between trend indicators of the second suite from *IndiSeas*-phase II. We also observed a high proportion of significant correlations between Trophic Level of the catch and Trophic Index (MTI). A high proportion of positive and significant correlations were also found between Predators and Trophic Level of the Community, Biomass and Trophic Level of the Model, Life Span and Trophic Level of the community, and IVI and Trophic Level of the catch. The proportion of negative and significant correlations between trend indicators was less than 50% in any case.

Considering previous results, some indicators could therefore be excluded from our ultimate list of indicators when assessing the status of exploited marine ecosystems: (i) Life Span, because it is correlated strongly with Predators both with regard to current state and trend indicators, and because Predators is deemed a more certain indicator since it does not rely on what are sometimes poor estimates of life span per species; (ii) the Trophic Level

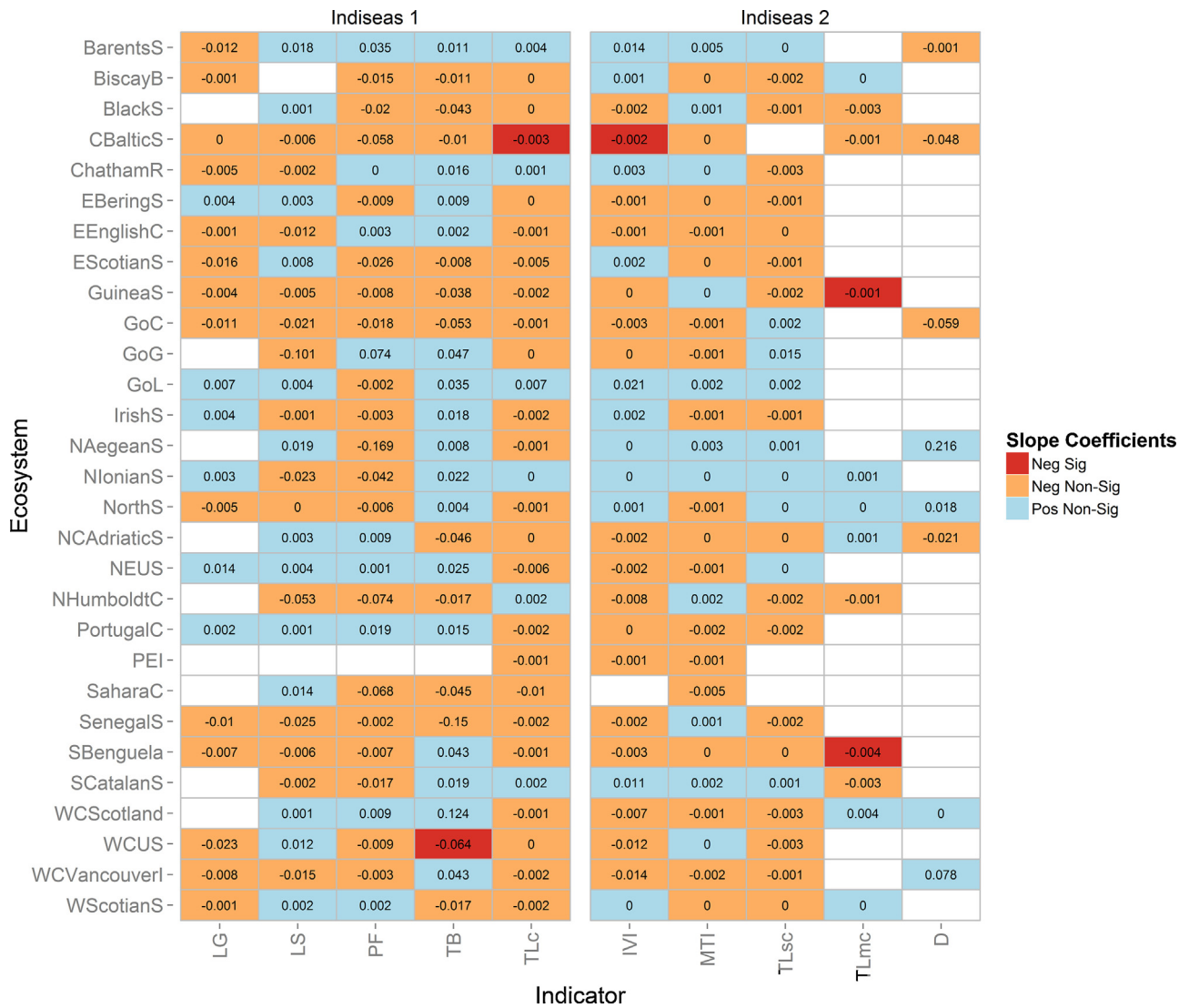


Fig. 4. Heatmap of trend indicators' slope coefficients (1980–2010, Figure S7) using both *IndiSeas*-phase I (left panel) and II (right panel) indicators (Table 1) and the estimation of rates of change over time method (value shown in cell; analysis allowed for non-linear changes over time). Neg: negative, Pos: positive, Sig: significant, Non-Sig: non-significant trend. Full indicator names and acronyms are listed in Table 1 and ecosystem names and labels are listed in Table 2.

of the Model, because there are strong correlations with Predators and Biomass in current state and trend indicators, respectively, and because models are available only for a small number of ecosystems; and (iii) the Landings/Discards indicator, which was difficult to estimate for several ecosystems and showed redundancy with

Sustainable Stocks. Finally, (iv) the Fish Size indicator should be considered carefully because of lower data availability and a high percentage of positive correlations with relative fishing effort (contrary to the expected decline in fish size with increasing fishing pressure; results presented in Section 3.5).

Table 3

Spearman's non-parametric rank order correlation coefficients (values below the diagonal) and associated *p*-values (values above the diagonal) of state indicators for the 29 exploited marine ecosystems (*n* values included in the analysis are: BS = 27, LG = 20, LS = 26, PF = 27, SS = 27, TLc = 29, MTI = 28, NDES = 22, TLsc = 24, TLmc = 12, D = 9). Significant correlations are highlighted in bold.

<i>p</i> -Value R	<i>IndiSeas</i> -phase I indicators						<i>IndiSeas</i> -phase II indicators				
	BS	LG	LS	PF	SS	TLc	NDES	MTI	TLsc	TLmc	D
BS		0.77	0.84	0.13	0.59	0.80	0.69	0.64	0.98	0.25	0.55
LG	0.07		0.06	0.08	0.66	0.04	0.69	0.26	0.20	0.70	0.13
LS	−0.04	0.44		0.04	0.00	0.46	0.44	0.13	0.07	0.49	0.18
PF	−0.30	0.40	0.41		0.40	0.57	0.91	0.37	0.37	0.00	0.58
SS	0.11	−0.11	0.56	0.17		0.16	0.07	0.90	0.30	0.51	0.01
TLc	−0.05	0.47	0.15	0.11	−0.28		0.98	0.00	0.62	0.06	0.19
NDES	−0.09	0.10	0.18	−0.03	0.40	0.00		0.06	0.56	0.09	0.30
MTI	0.10	0.27	0.31	0.18	−0.03	0.54	0.80		0.93	0.66	0.61
TLsc	0.00	0.31	0.39	0.19	0.22	0.11	0.14	−0.02		0.85	0.44
TLmc	−0.41	−0.20	0.27	0.86	−0.25	0.58	−0.57	−0.16	0.08		0.44
D	0.23	0.77	0.49	0.21	0.82	0.49	0.39	−0.21	0.72	−0.35	

Table 4
Proportion of negative and positive significant Spearman's non-parametric rank order correlation coefficients of trend indicators for the 29 exploited marine ecosystems (values below the diagonal; negative and positive values separated by a semicolon). The proportions are calculated taking into account the number of time series available in each ecosystem (values above the diagonal). Bold values highlight instances where the proportion of positive correlations between two indicators is more than 40%.

Prop. –ve, +ve correlations	IndiSeas-phase I indicators					IndiSeas-phase II indicators				
	LG	LS	PF	TB	TLc	IVI	MTI	TLsc	TLmc	D
LG										
LS	0.05; 0.21									
PF	0.1; 0.35	0.04; 0.57								
TB	0.05; 0.15	0.26; 0.19	0.29; 0.11							
TLc	0; 0.25	0.19; 0.19	0.18; 0.18	0.07; 0.14						
IVI	0.05; 0.1	0; 0.19	0.07; 0.11	0.04; 0.11	0.14; 0.54					
MTI	0; 0.2	0.15; 0.11	0.11; 0.14	0.17; 0.14	0; 0.66	0.18; 0.36				
TLsc	0.05; 0.32	0.16; 0.48	0; 0.50	0.15; 0.12	0.04; 0.19	0.04; 0.12	0.04; 0.19			
TLmc	0.14; 0.14	0.17; 0.25	0.15; 0.23	0.08; 0.58	0; 0.25	0.18; 0.09	0.25; 0.25	0.09; 0.27		
D	0; 0	0.13; 0	0.25; 0	0; 0.25	0.13; 0.13	0.13; 0	0.13; 0	0; 0	0; 0	

3.4. Status of exploited marine ecosystems

When comparing the status of exploited marine ecosystems using current state indicators from *IndiSeas*-phase I with the whole suite of indicators, we observed that the classification of ecosystems using multivariate techniques (cluster analysis and nMDS ordinations) varied significantly (Fig. 5). Using *IndiSeas*-phase I

state indicators, three groups of ecosystems emerged: the north Aegean Sea and the northeast U.S. emerged as different from the other ecosystems, which clustered together in a large group (Fig. 5a and c). Using the whole suite of state indicators, all the ecosystems clustered together and we did not discern any significant pattern (Fig. 5b and d). It should be noted that when the whole suite of indicators was used, the stress value in the nMDS ordination increased

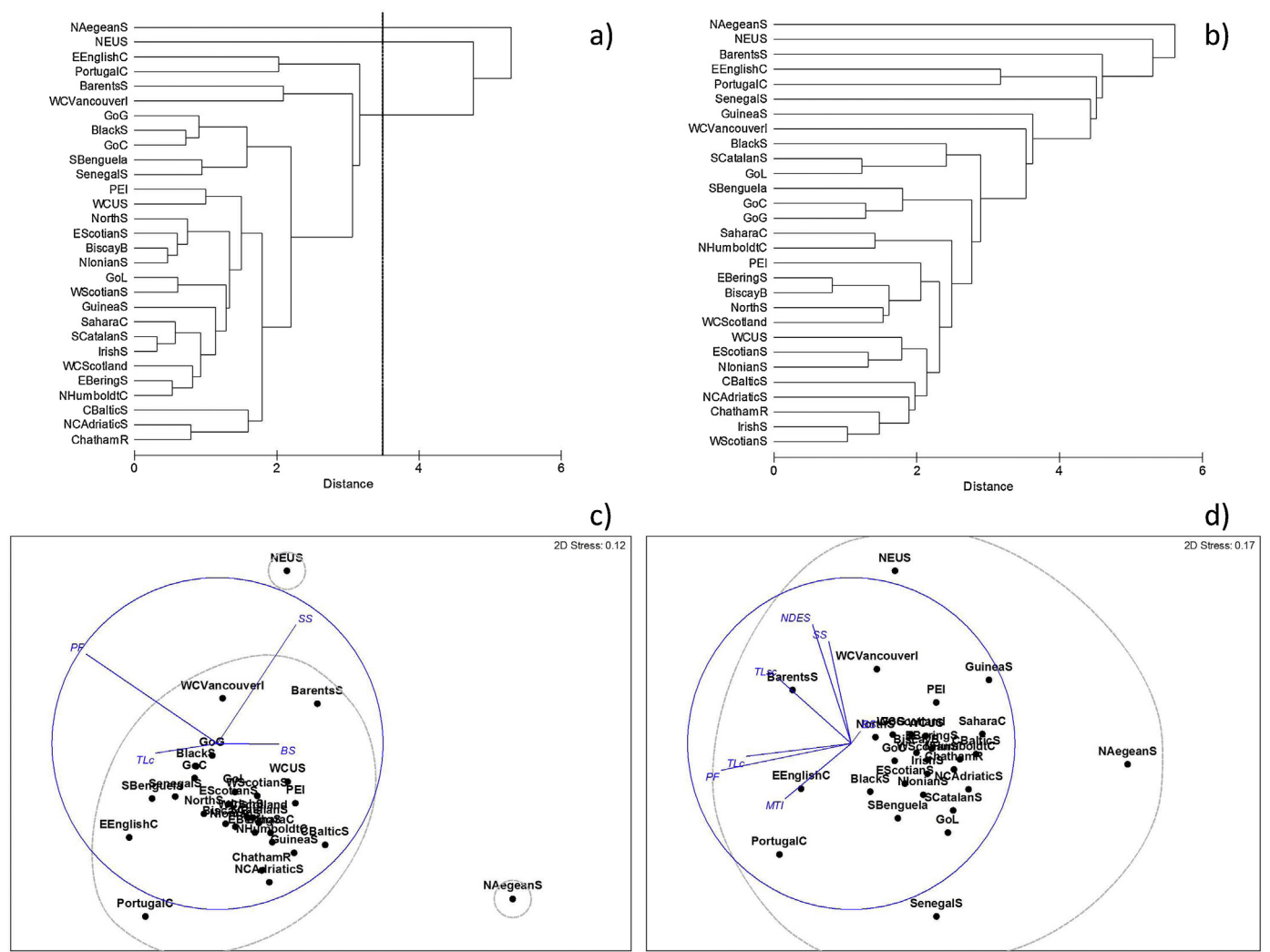


Fig. 5. Cross-comparison of current states (2005–2010) of ecosystems using cluster and non-metric MultiDimensional Scaling (nMDS) analysis with indicators from (a–c) *IndiSeas*-phase I, and (b–d) whole suite of indicators (excluding Life Span, Fish Size, Trophic Level of the Model and Landings/Discards). The Spearman's non-parametric rank order correlation contributions of each indicator are shown as vectors in the nMDS.

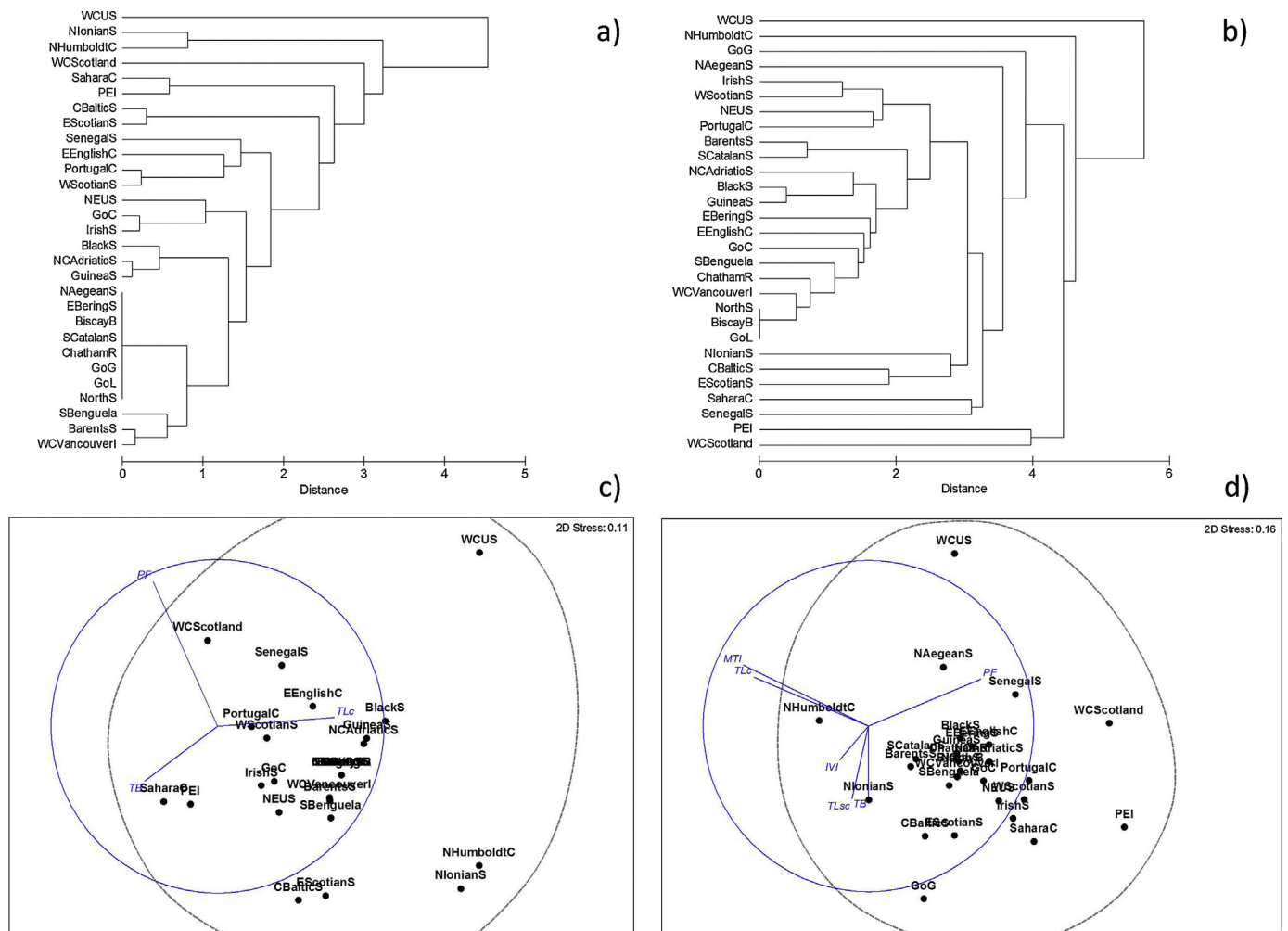


Fig. 6. Cross-comparison of trends (1980–2010, Figure S7) of ecosystems using cluster and non-metric MultiDimensional Scaling (nMDS) analysis with indicators from (a–c) *IndiSeas*-phase I, and (b–d) whole suite of indicators (excluding Life Span, Fish Size, Trophic Level of the Model and Landings/Discards). The Spearman's non-parametric rank order correlation contributions of each indicator are shown as vectors in the nMDS.

(from 0.12 to 0.17). This moderately high stress value indicates the difficulty in displaying the relationships, which generally suggests a loss of information when projecting from high dimension to two dimensions, when more indicators are incorporated. These indicators brought additional dimensions of similarity differences among ecosystems.

When comparing the status of exploited marine ecosystems using *IndiSeas*-phase I trend indicators resulting from the generalized least-squares analyses results, no different groups were observed in the classification of ecosystems (cluster analysis and nMDS ordinations) (Fig. 6c and d). However, the clustering of ecosystems was qualitatively different than when using the whole suite of trend indicators (Fig. 6a and b). When the whole suite of indicators was used, the stress value in the nMDS ordination also increased (from 0.11 to 0.16).

Due to redundancy of some indicators and/or poor availability of data as described above, all the above analyses were performed without Life Span, Fish Size, Trophic Level of the Model and Landings/Discards.

3.5. Correlations with fishing pressure

Since the indicators were formulated to decrease with higher fishing pressure (using relative fishing effort and mortality as

proxies), we expected a high proportion of negative correlations between the three measures of fishing pressure (Landings/Biomass, relative fishing effort and relative mortality) and the indicators.

The highest proportions of ecosystems with negative correlations were between Biomass and the fishing pressure indicator, Landings/Biomass (0.79; Table 5), which is logical due to the formulation of the pressure indicator. Among other indicators, proportions of significant positive or negative correlations with Landings/Biomass were less than or equal to 0.33. Among ecosystems with available relative fishing effort, the highest proportion of negative correlations were between Landings/Discards and relative fishing effort (0.50; Table 6a) and between Trophic Level of the Model and relative fishing effort (0.43; Table 6a). In contrast, 50% of the ecosystems showed a positive correlation between Fish Size and relative fishing effort, although this information was only available for four ecosystems. The rest of the indicators showed variable proportions (<0.29) of significant positive or negative correlations with relative effort. Among those ecosystems with available relative fishing mortality data, we observed that Biomass showed the highest proportion of ecological indicators with significant and negative correlations with relative fishing mortality (0.44; Table 6b). The rest of indicators showed variable proportions (<0.33) of significant positive or negative correlations with relative fishing mortality.

Table 5

Number and proportion of Spearman's non-parametric rank order correlation coefficients between *IndiSeas* indicators and the Landings over Biomass ratio indicator. The number per indicator provides the number of ecosystems with data available to test this relationship. Bold values highlight instances where the proportion of positive correlations between two indicators is more than 40%.

Landings/biomass	<i>IndiSeas</i> -phase I indicators					<i>IndiSeas</i> -phase II indicators				
	LG	LS	PF	TB	TLc	IVI	MTI	TLsc	TLmc	D
–ve, +ve correlations	3, 2	5, 6	4, 8	23, 0	9, 6	8, 3	6, 8	0, 4	4, 2	1, 0
–ve correlations (prop.) [*]	0.15	0.19	0.14	0.79	0.31	0.29	0.21	0.00	0.33	0.13
+ve correlations (prop.) [*]	0.10	0.22	0.29	0.00	0.21	0.11	0.28	0.15	0.17	0.00
Number per indicator	20	27	28	29	29	28	29	26	12	8

^{*} Proportion of significant correlations calculated with the number per indicator. –ve: negative; +ve: positive.

4. Discussion

4.1. *IndiSeas* ecological state and trend indicators

In this study we developed an analysis to evaluate a suite of current state and trends of ecological indicators to determine the status of 29 exploited marine ecosystems. We considered several ecological indicators that were defined to measure fishing impacts on commercial stocks, and capture the broader effects of fishing on marine biodiversity and ecosystems, some of which have important conservation implications.

Overall, our results illustrate that the two suites of indicators, *IndiSeas*-phase I and phase II, are often complementary and in some cases offer additional interpretations or information. Thus, the new suite of indicators selected to capture the broader effects of fishing on marine biodiversity and ecosystems provided additional information to complement that obtained by using only the first suite of *IndiSeas*-phase I indicators. Our study also highlights that the interpretation of indicators is complex because they show a diverse range of responses to fishing pressure and they require careful analyses and background knowledge of the ecosystems.

The first suite of ecological indicators chosen during *IndiSeas*-phase I (Shin and Shannon, 2010) were selected specifically to measure ecosystem response to fishing pressure, and have greater availability in terms of temporal and spatial coverage in the 29 case studies than the new indicators, chosen to capture aspects of the impacts of fishing on biodiversity. This is logical since the conceptualization and development of indicators for measuring the effects of fishing pressure on the exploited part of the community has been studied for a longer period of time (Rochet and Trenkel, 2003). In contrast, biodiversity issues have more recently been added to the analyses as the Ecosystem-based Approach to Fisheries and comprehensive evaluations of marine ecosystems have been gaining momentum (Halpern et al., 2012; Pikitch et al., 2004; Tittensor et al., 2014).

The additional suite of *IndiSeas*-phase II indicators was also available in many of our study systems, with the exception of the two extra indicators (Trophic Level of the Model and Landings/Discards). This is a positive result in the drive to achieve current and future targets dictated by international and regional frameworks, such as the Marine Strategy Framework Directive (MSFD) of the European Commission or the Aichi Targets of the Convention of Biological Diversity. The latest global evaluation of the Aichi Targets of the CDB only includes two indicators that can be used to explicitly inform Aichi Target 6, which evaluates the aim to manage marine ecosystems, sustainably avoiding adverse impacts on commercial and non-commercial species and habitats (Tittensor et al., 2014).

Our results also show that some redundancy between indicators exists and highlight the potential to remove a few indicators from our initial suite in order to reduce monitoring and data collection efforts. Regarding the *IndiSeas*-phase I suite (Shin et al., 2010b), the Life Span indicator could be removed in some ecosystems where it shows a redundancy with Predators. In addition, the Fish Size indicator was not always available and showed positive correlations with higher fishing effort in some cases, which may be counter-intuitive given the original rationale for the selection of the indicator. This is an interesting result that needs further investigation; for example, the Fish Size indicator may be capturing environmental influences through the level of fish recruitment or the success of size-based fishing limits in some regions, whereas in other highly degraded ecosystems its sensitivity to further heavier fishing may be limited. In addition, Fish Size and Trophic Level are highly correlated in several systems, which may highlight that size-based and trophic-based phenomena in some exploited fish communities can follow similar directions of change at the community level, as previously suggested (Jennings et al., 2001). However, Fish Size reflects important ecosystem functioning issues relevant, for instance, within the MSFD framework by involving at least Descriptor 3 on Populations of Commercially Exploited Species and Descriptor 4 on Food Webs (EC, 2008, 2010).

Table 6

Number and proportion of Spearman's non-parametric rank order correlation coefficients between *IndiSeas* indicators and (a) fishing effort and (b) fishing mortality time series. The number per indicator provides the number of ecosystems with data available to test this relationship. Bold values highlight instances where the proportion of positive correlations between two indicators is more than 40%.

	<i>IndiSeas</i> -phase I indicators					<i>IndiSeas</i> -phase II indicators				
	LG	LS	PF	TB	TLc	IVI	MTI	TLsc	TLmc	D
(a) Fishing effort										
–ve, +ve correlations	0, 2	2, 1	2, 1	2, 1	0, 2	1, 1	0, 2	0, 2	3, 2	1, 0
–ve correlations (prop.) [*]	0.00	0.29	0.29	0.29	0.00	0.14	0.00	0.00	0.43	0.50
+ve correlations (prop.) [*]	0.50	0.14	0.14	0.14	0.29	0.14	0.29	0.29	0.29	0.00
Number per indicator	4	7	7	7	7	7	7	7	7	2
(b) Fishing mortality										
–ve, +ve correlations	0, 1	2, 2	3, 2	4, 0	0, 3	1, 2	1, 2	1, 2	3, 2	1, 0
–ve correlations (prop.) [*]	0.00	0.22	0.33	0.44	0.00	0.13	0.11	0.11	0.33	0.33
+ve correlations (prop.) [*]	0.20	0.22	0.22	0.00	0.33	0.25	0.22	0.22	0.22	0.00
Number per indicator	5	9	9	9	9	8	9	9	9	3

^{*} Proportion of significant correlations calculated with the number per indicator. –ve: negative; +ve: positive.

Data to compute both extra indicators (Trophic Level of the Model and Landings/Discards) were not readily available. Landings/Discards data are missing from several ecosystems due to a general lack of surveys or monitoring programmes to register discarding practices in marine ecosystems (Kelleher, 2005). This points to a real problem when managing exploited marine resources and reinforces the fact that greater investment is needed to retrieve information about discarding, as the EC has recently highlighted in the new CFP requirements (Sarda et al., 2015). The deficiency of data available to calculate the Trophic Level of the Model reflects the absence of ecological models in many marine ecosystems, despite concerted efforts to develop these new analytical tools (Heymans et al., 2014). Therefore, more efforts should be geared towards developing ecosystem models to characterize the historic dynamics of marine ecosystems. Both extra indicators showed high proportions of negative correlations with fishing effort (a desirable trait in our selection of indicators), but also showed high correlations with other ecological indicators. Therefore, the omission of these two indicators should not substantially affect the assessment of the status of exploited marine ecosystems.

Our study considered four trophic level-based indicators. In a previous study, an extensive evaluation was undertaken of a variety of trophic level indicators across nine well-studied marine ecosystems using model, survey and catch-based trophic level indicators (Shannon et al., 2014). Results highlighted that the differences observed between trophic level indicator values and trends depended on the data source and the minimum trophic level included in the calculations, and where not attributable to an intrinsic problem with these indicators. Moreover, the exploitation history (in time and space) and the implementation of fisheries management measures in an ecosystem can influence what we can readily deduce from trophic level-based indicators. Therefore, these factors should be taken into account when using and interpreting trophic level-based indicators (Shannon et al., 2014). Still, the study concluded that all three types of trophic level indicators (i.e., catch-based, survey-based, and model-based) provide information that is useful for an EAF. Overall, our study supports these results.

Additionally, Shannon et al. (2014) found that catch-based trophic level indicators did not necessarily reflect what is happening at the community or ecosystem level since non-targeted and discarded or unreported species may not be considered. Catch-based indicators are intrinsically linked to fishing pressure and respond sensitively to management action but are not specific indicators of change in ecosystem state. Importantly, they often cover a longer period of time and provide a measure of the spread of pressure across trophic levels (Shannon et al., 2014). In our study, positive correlations were identified between Trophic Level of the Catch and the Trophic Index (MTI), which was expected since both are catch-based, but measure changes in all captured species versus only higher trophic-level species, respectively. Thus we suggest selecting just one of these two indicators from our suite if a shorter list is needed. The selection between the two should consider the species one wishes to include in the analysis (e.g. including or excluding small pelagic fish, invertebrates, etc.). For example, in global comparisons, in order to accommodate ecosystems in which low TL species dominate catches or at least catch variability (e.g. upwelling systems, Mediterranean systems) (Shannon et al., 2010), the use of Trophic Level of the Catch instead of the MTI is recommended. In upwelling systems, it is advisable to also use the Marine Trophic Index with cut-off at TL 4.0 in order to examine changes within the apex predator community while excluding small and medium pelagic fish, some of which have TLs above 3.25 (e.g. Peruvian anchoveta and South African anchovy *Engraulis capensis*)

and which are subject to large natural fluctuations in abundance (Shannon et al., 2010).

Survey-based trophic level indicators provide a fuller picture of what is happening at the community level and may capture combined effects of fishing and the environment more clearly, but are nonetheless also a limited information source given that they are based on a subset of those species present and often of limited temporal scope (i.e., only conducted over a short time), especially where only part of the ecosystem is surveyed (Shannon et al., 2014). Survey gears, such as trawls, are highly selective and available survey data from most ecosystems will have been collected using a restricted number of gear types so although the inclusion of the phase-II indicators does provide additional information in regard to the wider biodiversity, it is still an incomplete view of the true ecosystem state. In our study, the Trophic Level of the Community was useful to highlight specific processes in ecosystems as it was not redundant (i.e., low proportion of positive correlations with other, non-TL-based indicators) and was highly correlated with relative fishing pressure. Therefore, our study supports previous results suggesting that community-based indicators represent fishing impacts at the whole ecosystem level and should be incorporated where possible, as a means of providing additional information and improving understanding of ecosystem dynamics (Shannon et al., 2014), although data availability may be limiting especially in the case of modelled community indicators.

Furthermore, a separate study specifically looked at the Non-Declining Exploited Species (NDES) indicator and used it to compare patterns in the states and temporal trajectories of the exploited species of the community relative to the overall community (Kleisner et al., 2015). The NDES indicator was then compared with the Trophic Level of the Community, Predators, and Life Span. The study highlighted that in some ecosystems, the current states of the NDES indicator were consistent with other indicators, indicating deteriorating conditions in both the exploited portion of the community and the overall community. However differences in some ecosystems illustrated the necessity of using a variety of ecological indicators to reflect different facets of the status of the ecosystem. This is reinforced with our analysis, where a clear redundancy of the NDES indicator with the rest of selected indicators was not identified. Nevertheless, as is the case for other indicators, using the NDES indicator requires context-specific supporting information in order to provide guidance within a EAF management framework (Kleisner et al., 2015).

4.2. Cross-comparison of indicators to inform on the exploited marine ecosystem status

In general, both *IndiSeas*-phase I and phase II indicators confirm that mixed signals are common in many marine ecosystems when evaluating their status (Bundy et al., 2012). Thus, the cross-comparison of indicators to inform on the status of exploited marine ecosystems has been highlighted as an important practice in previous studies to avoid biases of specific indicators and blind interpretations (Bundy et al., 2012; Coll et al., 2010b; Kleisner et al., 2013). This study illustrates the insights gained in using a suite of ecological indicators, which can provide diverse information, but also highlights the complexity in understanding and interpreting the signals and driving mechanisms behind the responses.

The responses of indicators to pressures, in this case fishing, are not always linear and may be difficult to interpret because the indicators of fishing effort are not ideal proxies of fishing pressure or because the ecological indicators are responding to other extrinsic factors, such as environmental variables. Parallel results developed within the *IndiSeas* framework suggest that ecological indicators are in fact sensitive to environmental drivers (Fu et al.,

2015), which highlights that interactions between the indicators and at least one other extrinsic factor is likely. In addition, analyses of indicators assuming a linear relationship between response indicators and pressure indicators may be too simplistic. In fact, recent comprehensive studies of exploited marine ecosystems suggest that detailed information about past and present exploitation strategies, main productivity mechanisms, and dominant ecological and environmental traits are essential elements to correctly interpret ecological indicators to determine the status of exploited marine ecosystems (Fu et al., 2015; Kleisner et al., 2014; Link et al., 2010; Shannon et al., 2014; Shannon et al., 2010). This emphasises the need to investigate the sensitivity and specificity of indicators to different individual pressures, as well as multiple-interacting pressures, and their responsiveness to management thresholds and reference points (Large et al., 2013, 2015; Shin et al., 2012).

In this study, we focused on the effects of fishing, which is a major pressure in many ecosystems (Costello et al., 2010). Thus, the indicators were defined to decrease with greater fishing impact. However, it is important to recognize that fishing impact is not always the leading driver in an ecosystem, even in exploited ecosystems, and that other drivers, such as the environmental stressors, can have significant effects on indicators (Link et al., 2010; Mackinson et al., 2009). For example, in the Southern Benguela, the effects of fishing are confounded with ecosystem changes at least partially due to environmentally induced shifts in the distribution of key resources (Shannon et al., 2010, 2014). This has important implications for birds or mammals, which are often of conservation concern and also support tourism industries (e.g., Blamey et al., 2015). Like the Southern Benguela, Senegal and Guinea are ecosystems in which fish communities and landings are dominated by small pelagic stocks, thus the effects of fishing are probably confounded with ecosystem changes due to environmentally induced shifts, influencing the abundance and distribution of these key resources (Chavance et al., 2004; Roy et al., 2002). In the northern Humboldt Current anchovy is dominant when the ecosystem is considered healthy, the impact of indicators is a decreased mean fish length in the surveyed community, shortened mean maximum life span of surveyed fish species and reduced mean intrinsic vulnerability index of the fish landed catch, so that a decrease in these indicators is not always related to greater fishing impact (Chavez et al., 2008). In the west coast U.S. ecosystem, management actions have recovered many harvested species, but survey-based indicators declined over the period observed (2003–2010), coincident with 4–5 years of a warm, unproductive phase of the Pacific Decadal Oscillation and attenuation of a strong 1999 groundfish cohort (Keller et al., 2012; Tolimieri et al., 2013).

The importance of environmental drivers is also seen in the North Ionian Sea, where extensive fishing pressure and environmental shifts have had negative implications for short-beaked common dolphins (Piroddi et al., 2011). In the Portuguese Coast, environmentally induced shifts have also occurred (Borges et al., 2010), while important alterations have taken place in the central Baltic Sea ecosystem due to climate and multiple human induced impacts (Möllmann et al., 2009; Österblom et al., 2007). In addition, and compared to the other ecosystems, the relatively lower current state calculated for the Black Sea may be due to the dominance of small pelagic fish in this ecosystem and their strong fluctuation in landings due to nutrient enrichment, overexploitation and environmental change (Oguz et al., 2012). In the Barents Sea, the rapid fluctuations in stock size and landings due to natural drivers, in addition to fisheries regulations, have led to under-exploitation of long-lived species and increased landings of short-lived pelagic species in the presence good recruitment classes (Johannessen et al., 2012). Therefore, as has been previously recognized (Shannon et al., 2014; Shannon et al., 2010), detailed knowledge about the ecosystem is important to facilitate understanding of the

patterns revealed by the selected indicators. The influence of other drivers on ecosystems suggests that there is the need to consider additional ecosystem-specific indicators, such as environmentally linked response indicators (Boldt et al., 2014).

Despite these mixed signals, which in themselves should convey a need for cautious monitoring of future ecosystem conditions and trajectories, some ecosystems analyzed in this study are likely more impacted than others. Overall poor ecosystem status compared to other ecosystems considered can be described across the suite of indicators for several case studies, e.g., the Black Sea, the Gulf of Cadiz, the north Aegean Sea, the north Ionian Sea, the north-central Adriatic Sea, the northern Humboldt Current, the Sahara Coastal, the Senegalese shelf and the southern Catalan Sea if considering current states, and central Baltic Sea if considering trends. This is in line with information from the literature (e.g., Coll et al., 2008, 2010a; Gascuel et al., 2015; Piroddi et al., 2010; Torres et al., 2013). Therefore, this study illustrates that several exploited marine ecosystems have a relatively high impact by fishing, in line with previous studies (Bundy et al., 2010; Coll et al., 2010b; Kleisner et al., 2013; Shannon et al., 2010) and highlights the need to develop improved management tools considering conservation issues of natural resources.

In addition, our results show important differences for how ecosystems are classified using current state and trend indicators when explicitly considering the impacts of fishing on biodiversity. Indicators that capture the dynamics of the fuller spectrum of fish within an ecosystem, such as Trophic level of the Community and the Non-Declining Exploited Species indicator, convey additional information that complement that already provided by the more traditionally accepted suite of ecological indicators used for detecting fishing impacts, and can serve to strengthen the signals we may be receiving as warning of impending ecosystem change. Thus, the new suite of *IndiSeas*-phase II ecological indicators provide useful additional information in relation to wider biodiversity aspects of the effects of fishing and highlight the potential for other factors that should be considered when evaluating ecosystem status. In systems where the patterns in the old and new suites of indicators are similar, they may still provide extra context and support for the patterns seen with the *IndiSeas*-phase I indicators. These indicators should be considered complementary to other ecological indicators that measure fishing impacts on commercial stocks and communities by capturing the broader effects of fishing on marine biodiversity and ecosystems. While this study focusses specifically on the effects of fisheries, wider ecosystem assessments of other drivers in the marine environment (e.g. marine tourism, mining and aquaculture) may also benefit from inclusion of a wider range of biodiversity and conservation-based indicators.

In a world largely focussed on exploited species, it seems that indicators that capture the broader effects of fishing on marine biodiversity help move towards the conciliation of exploitation and conservation issues (Brander, 2010; Palumbi et al., 2008; Worm et al., 2006). Thus, biodiversity and conservation-based indicators should be used in concert to provide additional useful information to evaluate the overall impact of fishing on exploited marine ecosystems.

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