Fish and Fisheries
March 2016, Volume 17, Issue 1, Pages 31-55
http://dx.doi.org/10.1111/faf.12090

Fishing impact and environmental status in European seas: a diagnosis from stock assessments and ecosystem indicators

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Abstract:

Stock-based and ecosystem-based indicators are used to provide a new diagnosis of the fishing impact and environmental status of European seas. In the seven European marine ecosystems covering the Baltic and the North-east Atlantic, (i) trends in landings since 1950 were examined; (ii) syntheses of the status and trends in fish stocks were consolidated at the ecosystem level; and (iii) trends in ecosystem indicators based on landings and surveys were analysed. We show that yields began to decrease everywhere (except in the Baltic) from the mid-1970s, as a result of the over-exploitation of some major stocks. Fishermen adapted by increasing fishing effort and exploiting a wider part of the ecosystems. This was insufficient to compensate for the decrease in abundance of many stocks, and total landings have halved over the last 30 years. The highest fishing impact took place in the late 1990s, with a clear decrease in stock-based and ecosystem indicators. In particular, trophic-based indicators exhibited a continuous decreasing trend in almost all ecosystems. Over the past decade, a decrease in fishing pressure has been observed, the mean fishing mortality rate of assessed stocks being almost halved in all the considered ecosystems, but no clear recovery in the biomass and ecosystem indicators is yet apparent. In addition, the mean recruitment index was shown to decrease by around 50% in all ecosystems (except the Baltic). We conclude that building this kind of diagnosis is a key step on the path to implementing an ecosystem approach to fisheries management.

Keywords: Ecosystem approach to fisheries management ; ecosystem indicators ; good environmental status ; Marine Strategy Framework Directive ; stock assessment ; trophic level

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1. Introduction

Since the publication of the Code of Conduct for Responsible Fisheries (FAO 1995), the ecosystem approach to fisheries management (EAFM) has progressively been recognized as a necessity worldwide. The concept aims at assessing the global impact of fisheries on ecosystem functioning, whilst taking into account the fact that fisheries are embedded into the environment and cannot be managed in isolation (Garcia et al. 2003; Jennings and Rice 2011; Pikitch et al. 2004; Rice 2011). More generally, the ecosystem approach to fisheries management is the application of sustainable development principles to the fishing sector, combining ecological sustainability of stocks and ecosystems, economic viability of the fishing industry, and social viability and fairness for local communities as well as the broader society (Garcia and Cochrane 2005; Gascuel et al. 2011a).

In the European Union, efforts to implement the ecosystem approach to fisheries management led to the definition of a reference list of nine ecosystem indicators (STECF 2006 & 2007; European Commission 2008a), with the objective of assessing the fisheries impact not only on targeted stocks, but also on fish communities, biodiversity, seafloor integrity, population genetics, discarded species, and fishery fuel consumption. In recent years, several authors and working groups under the auspices of the International Council for Exploration of the Sea (ICES) have used or developed these ecosystem indicators (Greenstreet et al. 2011 and 2012; Shephard et al. 2011; Fung et al. 2012). Nevertheless, analyses of ecosystem impacts of fisheries remained partial, covering only some ecosystems with no associated standard monitoring in place at the pan-European level.

Things have started to change recently with the implementation of the Marine Strategy Framework Directive (MSFD; European Commission 2008b) by the European Commission. The overarching goal of this Directive is achieving Good Environmental Status across all European marine waters by 2020, based on 11 qualitative descriptors (European Commission 2010) and a set of associated indicators that is still under development. While the ecosystem approach first emerged in the context of fisheries management, the MSFD now requires its implementation of the ecosystem approach in the wider context of integrated management involving multiple sectors beyond fisheries. The indicators initially proposed for ecosystem approach to fisheries management are now finding their way into the MSFD as both share common sustainability goals.

Several working groups have been set up by ICES and the European Commission to work on this. In particular, the European Scientific Technical and Economic Committee for Fisheries (STECF) has set up an expert working group on the “Development of the Ecosystem Approach to Fisheries Management in European Seas”, with the overall objective of developing a feasibility approach to provide useful advice on ecosystem status in support of the Common Fisheries Policy. In line with the MSFD implementation, one of the main objectives of this working group was to assess the health of European ecosystems, using currently available data.

The current paper presents the main 2013 results from this working group. Our aim is to show that stock-based and ecosystem-based indicators can provide a complementary diagnosis on the fishing impact and environmental status of European seas. Seven ecosystems covering the Baltic and Atlantic European marine waters (West Scotland and Ireland, Irish Sea, North Sea, Celtic Sea, Bay of Biscay, Iberian Coast) were used as case studies. For each of them: i) trends in landings since 1950 were examined with the objective of providing a comprehensive overview of the dynamics of the whole fishery; ii) integrated syntheses of the status and trends in fish stocks, derived from ICES assessments, were consolidated at the ecosystem level; and iii) trends in ecosystem indicators were analysed based on available time series of landings and scientific survey data.

Material and Methods

Marine ecosystems considered and data used

The current study considered seven European ecosystems covering the Baltic and Atlantic waters of the European seas (Fig. 1). These ecosystems refer to the reference list of marine ecosystems defined by the European Scientific Technical and Economic Committee for Fisheries (STECF, 2011) which have to be considered as the functional and assessment units used for the operational implementation of the Ecosystem approach to fisheries management in European waters. These STECF marine ecosystems are comparable with the MSFD (sub)regions, but, according to the marine eco-regions defined by ICES (2004), ecosystems boundaries have been defined in order to match the divisions or sub-divisions used for fisheries statistics and stock assessments.

A specific database was set up to compile the various tables required for the current analysis:

(i) The ICES Statlant database (www.ices.dk, acceded April 2013) was used to analyse trends in landings from 1950 to 2010. This international database of fisheries landings is coordinated by ICES and includes landings of fish and shellfish from 20 countries, at the spatial resolution of ICES divisions and subdivisions. Landings were aggregated by ecosystem according to the boundaries of the ecosystems analysed. Until 1982 some landings were reported for a pool of several ICES subdivisions (referring to ICES areas VII, VIII, or VIIde the English Channel). These landings were distributed among ecosystems proportionally to the mean landings of the two most recent decades. This allocation has negligible effects on catch, except in the Bay of Biscay. Therefore, landings-based indicators were not calculated prior to 1983 in the Bay of Biscay.

(ii) Data related to all stocks assessed by ICES were used in order to build stock-based aggregated indicators at the ecosystem level. Catches, spawning stock biomass (SSB), fishing mortality rate (F), recruitment (R), and reference values for F and SSB were extracted for the 2012 single stock assessments from the ICES website (www.ices.dk; accessed 30 April 2013). Until 2010, ICES used reference values for fishing mortality and spawning stock biomass based on the ‘precautionary approach’ as thresholds for sustainable exploitation (i.e. F_{pa} and B_{pa}, to determine if a stock is within “safe biological limits”, thus avoiding recruitment overfishing). According to the commitments of the 2002 World Summit of Sustainable Development (WSSD 2002, Johannesburg), the MSFD, as well as the revised Common Fisheries Policy, now aim for a sustainable exploitation and new reference values for fishing mortality, F_{MSY} (i.e. the fishing mortality assumed to produce the Maximum Sustainable Yield, thus avoiding growth overfishing), was adopted. More precisely the fishing mortality of overexploited stocks should be reduced to F_{MSY} by 2015 ‘where possible’, and for all stocks by 2020. As the previous reference value for fishing mortality (F_{pa}) was, in practice, often treated as a target value (Piet and Rice, 2004) while the current and markedly lower F_{MSY} should be treated as a limit, this should lead to a significant decrease in fishing mortality and a subsequent increase in SSB for almost all stocks. Therefore, the management of European fisheries is currently following a transition scheme, using the three reference values B_{pa}, F_{pa} and F_{MSY}. These values were extracted, when available, from the ICES website and used in the current analysis. For stocks not assessed in 2012, the last available assessment (from 2005 to 2011) was considered. A total of fifty-seven assessed stocks were included in the analyses (see the list in the Supplementary Material). When a stock occurred in several ecosystems, catches and biomass estimated from the assessment were allocated in each ecosystem in the same proportions as the mean 2000-2010 ratio of landings per ecosystem.

(iii) Trawl survey data were extracted from ICES DATRAS database (online DAtabase of TRAwl Surveys, datras.ices.dk; accessed 30 March 2013) to calculate ecosystem indicators along the longest standardized time series possible. For consistency, demersal trawl surveys with a similar protocol were selected for each ecosystem (for instance, the time series selected for the North Sea starts in 1983, the year in which all areas of the ICES International Bottom Trawl Survey were conducted with a standardized GOV trawl gear). Only surveys covering the larger part of the ecosystem were considered (i.e. local coastal surveys were excluded), using the stations located within the studied ecosystem (see details on surveys selection and data extraction in the Supplementary Material). In the Celtic Sea two surveys occurred each year (France-Evhoé and UK-WCGFS), using distinct sampling design, gears and vessels. In this case two different estimates were calculated for each indicator.

Landings and stock-based indicators

ICES Statlant statistics were used to analyse long-term trends in total landings of the seven ecosystems. Times series of two indices of the landed species diversity were calculated for each ecosystem, based on:

(i) The number of exploited species whose landings were significant, i.e. higher than a minimum level, conventionally set equal to 0.5% of the mean annual total landings of the last ten years (i.e. 2001-2010);

(ii) The Shannon diversity index (Shannon 1948): $H' = \Sigma_s [P_s \cdot \log_2(P_s)]$, where P_s is the proportion in mass of species *s* in the yearly total landings.

The proportion of exploited species covered by stock assessments was computed for the 1950-2010 period. It reflects current assessment-based knowledge about the fishable fraction of the ecosystem. Then, for all stocks subjected to a stock assessment, three indicators were estimated to produce a synthesis of multiple stock trajectories at the ecosystem level: the total spawning stock biomass, the mean fishing mortality, and the mean recruitment index. Recruitment indices were computed per stock as the ratio of recruitment in year *y* divided by the average recruitment of that stock over the period where data for all species was available. The mean fishing mortality and the mean recruitment index were then averaged over the number of species using a geometric mean.

For many ecosystems, there was only a relatively short period during which data were available for all assessed stocks. By restricting the number of stocks included in indicator calculations, longer (but less representative) time series may be built. Therefore, within each ecosystem and for all ecosystems combined, two indicators were considered. The first one is related to all the currently assessed stocks. The second one is based on a subset of stocks, choosing a minimum of 60% of the assessed stocks allowing for the calculation of the longest possible but still sufficiently representative, time series. Sensitivity analyses to various subsets of the stocks considered are included in the Supplementary Material.

The current status of assessed stocks and their overall mean trajectory over time was summarized within each ecosystem on a common graph in reference to what are considered the main aspects of stock status, fishing mortality (F) and reproductive capacity (SSB). Two reference values were considered for the fishing mortality: the point at maximum sustainable yield (F_{MSY}) and the precautionary reference point (F_{pa}, which is in fact usually higher and thus less precautionary than

F_{MSY}). For SSB, only the precautionary biomass was used (B_{pa}) because B_{MSY} is currently not considered as a threshold for stock management in European waters and values are not available. For all stocks for which F_{MSY}, F_{pa} and B_{pa} limits were estimated by ICES, the comparison of the assessments (current F and SSB estimates) with associated reference points were presented following a modified version of the synoptic method developed by Garcia and de Leiva Moreno (2005). Thus, the current F is compared to reference points (here F_{MSY} and F_{pa}) by estimating a normalized index of fishing mortality as: $F^* = (F_{current} - F_{MSY})/(F_{pa} - F_{MSY})$. It should be stressed here that the resulting normalized fishing mortality index F* is no longer proportional to realized F, but rather it increases with F. F* is conventionally set equal to 0 and 1, for $F_{current}=F_{MSY}$ and $F_{current}=F_{pa}$ respectively. Thus, it allows us to simultaneously assess stock status in reference to both the 'old' F_{pa} and the 'new' F_{MSY} reference values. The normalized biomass only refers to B_{pa} and thus is expressed as $B^* = (B_{current})/(B_{pa})$. Trajectories in overall stock status were obtained calculating the mean F* and B* for each year (replacing $F_{current}$ by F_{year} in previous equations) and for all assessed stocks (for which target limits are known). These indicators F* and B* allow for the representation of the current status and the mean trajectory of assessed stocks in a single graph (e.g. Figures 5 and 6).

Ecosystem indicators calculated

Four ecosystem indicators were calculated from the survey data:

(i) The large fish indicator (LFI) reflects the size structure of the fish assemblage, which is assumed to be primarily affected by size-selective exploitation but is mediated by species composition (Shephard et al. 2012) as well as the fishing-induced reduction of life expectancy of each exploited species. This indicator was calculated as: $LFI = W_{>40cm} / W_{total}$, where $W_{>40cm}$ is the weight of fish greater than 40 cm in length and W_{total} is the total weight of all fish in the survey (Greenstreet et al. 2011; see details on calculations in the Supplementary material).

(ii) The mean maximum length of fish (MML) reflects the species composition of a fish assemblage, where fishing is expected to cause a decrease in the proportion of species with large asymptotic body size, slow growth rate, late age and large size at maturation (Shin et al. 2005). This indicator was calculated according to ICES (2009) based on the asymptotic total length of each species ($L\infty_s$ from Fishbase; Froese and Pauly 2012; www.fishbase.org; accessed 30 March 2013) as: $MML = \Sigma (W_s . L\infty_s) / \Sigma W_s$, where W_s is the total weight of species s caught during the survey.

(iii) The mean trophic level (MTL) of all fish caught during the survey indicates the effect of fishing on the food web (Jiming 1982; Pauly et al. 1998). It was calculated as: $MTL = \Sigma (TL_s . W_s) / \Sigma W_s$, where TL_s is the mean trophic level of species s (from Fishbase) and W_s is the total weight of species s caught during the survey.

(iv) The marine trophic index (MTI) reflects the trophic structure of the fish assemblage where fishing is expected to affect mostly the upper part of the food web, i.e. predatory fish. It is defined as the mean trophic level of predatory fish caught during each survey, taking into account only species whose trophic level is higher than or equal to 3.25 (Pauly and Watson 2005).

As such we refer to the large fish indicator (LFI) and mean maximum length (MML) as length-based indicators while the mean trophic level (MTL) and marine trophic index (MTI) are trophic indicators even though strictly speaking only the LFI captures changes in size structure while the three others reflect only changes in species composition, weighted by the species asymptotic total length (MML) or mean trophic level (MTL and MTI).

In order to highlight trends rather than the short term variability, all indicators were smoothed using a three year moving average. For the Celtic Sea, mean indicators were calculated by averaging estimates from the two available surveys (Evhoe and UK WCGFS). Mean indicators were also calculated for all ecosystems together except the Iberian coast, due to the very short time series available. Because the surveys selected in the various ecosystems did not cover the same time period, such a calculation required a preliminary standardisation of each time series. This standardization was obtained by rescaling the indicator series for each given ecosystem to the mean value of this indicator for all ecosystems over the 1997-2008 period, which is common to all the selected surveys.

Ecosystem indicators were also calculated from commercial fishery landings using the same equations (except LFI, because length frequencies were not available for landings). In this case, the mean trophic level (MTL) and the marine trophic index (MTI) were calculated for all species landed, including finfish and invertebrates. Trophic levels of invertebrates were extracted from SeaLifeBase (www.sealifebase.org; accessed 30 March 2013) or, when not available, conventionally assumed equal to 2.6 (Guénette and Gascuel 2012).

Trophic-based indicators were shown to be sensitive to the value of the trophic levels used for the top level species (Branch at al. 2010). Thus, sensitivity analyses were conducted, either changing the TL of cod (according to Branch et al. 2012), or using TLs from local Ecopath models. Detailed results of these analyses are presented in the Supplementary Materials and briefly summarized below. It should also be stressed that landings-based indicators are supposed to reflect ecosystem structure but may be biased by changes in fishing activities, involving gear selectivity (e.g. by technical creep or substitution of gear) or spatial distribution caused by the availability of quota. Thus, from a theoretical point of view, indicators based on surveys are preferred for un-biased analysis of fishing-induced changes in ecosystem health. However, because in practice, surveys only consider a subset of the fish community (i.e. often demersal finfish) and cover a relatively short period, complementary indicators based on landings can be applied to put the survey-based information in a longer-term and broader perspective.

Results

Long term trends in landings

Total landings in European seas increased from 3 million tonnes in the early 1950s to more than 7.2 million in the mid-1970s (Fig. 2). Since that period, landings have been decreasing, slowly until the mid-1990s but accelerating during the last period, falling to 4.3 million tonnes in 2010. In the North Sea, which is by far the most important fishing area in Europe, yield declined by >50% over this period, from almost 4 million tonnes in the turn of the 1970s, to around 1.7 million tonnes during the most recent years. The same trends were observed in almost all the considered European ecosystems, with landings peaking during the 1970s and strongly declining afterwards: from 160 000 to 60 000 tonnes in the Irish sea, from 780 000 to 450 000 tonnes in the Celtic Sea, from 350 000 to 130 000 tonnes in the Bay of Biscay and from 740 000 to less than 400 000 in the Iberian coast marine ecosystem.

Only two ecosystems exhibited a different pattern. In the Baltic Sea, landings were close to 1 million tonnes during the 1970s, peaked above this value in the mid-1990s, before slightly decreasing to less than 0.8 million during the last ten years. In the West of Scotland and Ireland, landings increased over almost the entire period, reaching a maximum of 1.4 million tonnes in 2006, before being halved in the most recent years. This particular trend was due to a single species, the blue whiting (*Micromesistius poutassou*, Gadidae), whose exploitation started in deeper waters in 1975, increased in the 1990s, reached 1.1 million tonnes in 2006 and subsequently declined. In this ecosystem, the total landings of other species followed a more common pattern with a maximum in the 1980s (around 700 000 tonnes) decreasing to about 50% at present.

The EU is by far the dominant fishing operator within European seas. In the considered ecosystems, more than 80% of the total landings were caught by EU Member States, landings by non-EU countries being significant only in the North Sea (mainly due to Norway), the West Scotland and Ireland (mainly due to Iceland), and the Baltic Sea prior to 1990 (mainly due to the former Soviet Union and Poland).

The share of landings from stocks assessed by ICES increased over the studied period, reaching approximately 90% of the total landings since the 1980s, in the three northern ecosystems: the Baltic Sea, the North Sea, and the West of Scotland and Ireland (note that, in certain years, catches used by ICES scientists were even greater than official catches from the ICES Statlant database, and that most ICES assessments considers discards and potential un-reported landings as well as potential re-allocation of landings between areas or between species). In contrast, stock assessments cover a smaller part of the total landings in the four southern ecosystems further decreasing over the most recent years, from about 60% in the 1980s or 1990s to around 40% today.

The decrease in catch observed in almost all the studied ecosystems over the last 3 or 4 decades occurred while new species started to be exploited intensively. This was especially the case for sandeels (*Ammodytes marinus*, Ammodytidae, more than 1 million tonnes in the North Sea), Norway pout (*Trisopterus esmarkii*, Gadidae), mackerel (*Scomber scombrus*, Scombridae), horse mackerel (*Trachurus trachurus*, Carangidae) and for some crustaceans and molluscs. More generally, the landings diversity indices show that a progressively greater proportion of each ecosystem was exploited over the period (Fig. 3). The number of species significantly exploited within each ecosystem peaked in the 1980s for the Irish Sea and West Scotland/Ireland ecosystems, in the 1990s for the Celtic Sea and the Iberian Coast ecosystems, and in the 2000s for the North Sea and the Bay of Biscay ecosystems. On average, the number of species significantly exploited jumped from 8 in 1950 to almost 24 in the late 1990s. At the same time, the Shannon diversity index (H) increased from 2.8 to 3.7. In other words, the increase in catch observed in the 1950s and 1960s progressively included more exploited species, and landings became more diverse. This process continued until the mid or late 1990s, while catches declined, suggesting fishermen tried to compensate for their losses by the exploitation of new resources. In the most recent years, the H diversity index remained high, while the number of exploited species slightly decreased, with landings of some species declining below the minimum level considered in the index calculation. The general pattern of an increasing diversity of catch until the 1990s is observed in all ecosystems, with the exception of the Baltic Sea, and to a lesser extent of the West of Scotland and Ireland. In the first case, diversity remained very low and decreased slightly since the early 1970s, while in the second case the predominance of the blue whiting starting in the 1980s reduced the diversity of landings.

Stock-based indicators

In the seven ecosystems considered, the fishing mortality index, reflecting mean fishing pressure on the assessed stocks, exhibited high values in the 1990s and a clear decreasing trend over the last 12 years. On average, for all the 57 available assessed stocks together, fishing mortality increased from 0.45 in the mid-1980s to almost 0.55 in 1998, and then decreased to approximately 0.30 by 2010 (Fig. 4 left column). Results appeared very consistent for all time series (i.e. either for short time series with all stocks or for longer time series based on fewer stocks). The same decreasing trend in fishing mortality was observed in all the studied ecosystems, with lower values around 0.25 at the end of the time series in the North Sea, the Celtic Sea and the Iberian coast. In the Baltic Sea, moderate fishing mortalities around 0.35 were already observed in the 1980s and the recent decrease is a return to this rather moderate level following a high fishing mortality period in the late 1990s. In all other ecosystems, fishing mortalities were lowest in the most recent years of the available time series. Note that in the Bay of Biscay several important stocks (cod *Gadus morhua*, Gadidae, anglerfish *Lophius* spp., Lophiidae) were not assessed for the most recent years, thereby leading to reduced knowledge over this period for this ecosystem.

Trends in stock abundances, based on spawning stock biomass time series (SSB), fluctuated between 10 and 14 million tonnes over the period, decreasing from the late 1980s to the mid-1990s, increasing until the mid-2000s and decreasing again in the most recent years (Fig. 4 middle column). This index was driven by a small number of large stocks, with contrasting trends between ecosystems. Blue whiting is the main stock driving changes observed in the West Scotland /Ireland ecosystem, with an SSB peaking in the mid-2000s before declining. Horse mackerel was especially abundant in the late 1980s, inducing an increase in the overall assessed SSB for the Celtic Sea and the Bay of Biscay. In the last ten years, when the fishing pressure was decreasing, an overall increase in SSB was only observed in the North Sea, essentially due to the recovery of plaice and herring. The Irish Sea, the Celtic Sea, and the Bay of Biscay exhibited an increasing trend in SSB while remaining at low levels compared to earlier periods. On the Iberian coast, SSB continued to decline reaching its lowest values at the end of the period.

In contrast, recruitment indices exhibited a consistently decreasing trend. On average for all assessed stocks together the mean recruitment has approximately halved since the mid-1980s (Fig. 4 right column). The same trend was observed with the same order of magnitude in all the studied ecosystems, with the only exception being the Baltic Sea, where recruitment fluctuated with no clear trends. In the North Sea, the West of Scotland and Ireland, and the Irish Sea the decrease occurred over a long period, apparently starting from the 1980s. In the Celtic Sea and the Bay of Biscay, the decline was only observed during the last decade.

Mean stock status and trajectories

ICES have estimated single stock based reference levels for 21 European assessed stocks (representing 34 % of the total 2010 landings). Among these, nine stocks met the current ICES management targets, with biomass above B_{pa} and a fishing mortality below the F_{MSY} level (Fig. 5). This is the case for plaice (*Pleuronectes platessa*, Pleuronectidae) in the North Sea and the Irish Sea, haddock (*Melanogrammus aeglefinus*, Gadidae) in the North Sea and Western waters (Rockall), saithe (*Pollachius virens* Gadidae) in the North Sea, Baltic herring (sub-div.30) (*Clupea harengus*, Clupeidae), and cod and sole (*Solea vulgaris*, Soleidae) in the Celtic sea. In contrast, eleven stocks failed to meet the requirements of the 'past' precautionary approach, with biomass lower than B_{pa} and/or fishing mortalities higher than F_{pa}. This especially applies for four strongly depleted stocks (i.e. biomass lower than $0.5*B_{pa}$): sole in the Irish Sea, and cod in the North Sea, Irish Sea and West Scotland and Ireland. The criterion for reproductive capacity B_{pa} was not met by three additional stocks (West of Scotland haddock, and the North Sea and Skagerrak stocks of sole), while four others exhibited fishing mortalities higher than F_{pa} (the Baltic cod (sub-div.22-24) and herring (riga), and the Biscay and the East Channel stocks of sole). Finally, one stock (West Channel plaice) met the requirements of the precautionary approach (F_{pa} and B_{pa}) but not the F_{MSY}

criterion. It can also be noted that among the 21 assessed stocks, a majority exhibited low biomass with 7 stocks below the precautionary level B_{pa} and 7 additional stocks close to that level (between 1.0 and 1.3 B_{pa}).

The trajectory of average fishing mortality of these assessed stocks confirmed that fishing mortalities were very high in the 1980s and 1990s, with mean values above F_{pa} (Fig. 6). Starting in the early 2000s, fishing mortality decreased and the mean value has been between F_{pa} and F_{MSY} since 2008, and very close to F_{MSY} in 2011 (but based on only 16 stock assessments). In spite of this, no increase was observed in the mean spawning biomass of these stocks which remains at a low level, close to B_{pa} and thus far below B_{MSY}. A similar trajectory was observed in the North Sea, with a current mean fishing mortality between F_{pa} and F_{MSY} and biomass decreasing over the most recent years and currently very close to B_{pa}. In the West of Scotland and Ireland and in the Celtic Sea, the F_{MSY} target was reached in 2011 (mean F below F_{MSY}), but the mean spawning biomass of assessed stocks remained very low, still decreasing and below B_{pa} in the former, slightly increasing and above B_{pa} in the latter. The three Irish Sea stocks showed mean fishing mortality fluctuating above F_{pa} while the mean SSB decreased to below B_{pa} from the late 1990s onwards and was still very close to that level in 2011. Once again, the Baltic Sea exhibited a different pattern, with mean fishing mortality fluctuating markedly, with moderate values occurring in the early 1990s, higher levels in the 2000s and current mean fishing mortality between F_{pa} and F_{MSY}.

Ecosystem indicators

Ecosystem indicators exhibited contrasting trends among ecosystems (Fig. 7).

(1) In the Baltic Sea, where the biodiversity is lower (Narayanaswamy et al. 2013), indicators based on demersal surveys were mainly driven by cod abundance. Thus, higher values observed for all indices between 1990 and 1996 may be attributed to a temporary increase in cod recruitment after the 1993 inflow event, while the increase occurring from about 2006 can be attributed to a recent overall increase in cod recruitment. Landings-based indicators provide a broader picture taking into

account demersal and pelagic species and a longer period starting in 1950. A clear and strong decreasing trend is observed over the whole period in the mean trophic level index (MTL went from 3.7 to 3.2) and in the mean maximum length (MML went from more than 70 cm to around 30 cm). This trend is mostly driven by a decrease in cod landings (representing almost 50 % of total landings in the 1950s and less than 10 % in the 2000s) and by the huge increase in sprat landings (from less than 5 % to more than 50 % of total landings). An increase in landings was also observed for some other low trophic level species, such as bivalves (e.g. mussel *Mytilus edulis*, Mytilidae, and common roach *Rutilus rutilus*, Cyprinidae*)*, while landings of whiting (*Merlangius merlangus*, Gadidae) - a high TL species - decreased over the period. Note that the high cod recruitment occurring in the early 1980s temporarily interrupted the long-term decrease of the two indices, which accelerates in the following years. In this ecosystem, landings having high TL comprise almost exclusively cod, explaining why the marine trophic index MTI was remarkably stable over the whole period.

(2) In the North Sea, length-based indicators from surveys (LFI and MML) decreased between 1985 and 1993 and have remained at a low level over the last twenty years, suggesting that the ecosystem is now dominated by small fish and other small species. The mean trophic level estimated from surveys decreased over the whole period (the MTL went from 4.1 to 3.9), but with a large year-to-year variability masking the trend over the most recent years. Indicator values based on landings decreased slightly during the 1970s and 1980s (the mean trophic level MTL went from 3.4 to 3.3, and the mean maximum length MML from 50 to 40 cm), at a time when a larger part of the ecosystem started to be exploited with an increasing catch of sandeel, mackerel and sprat (*Sprattus sprattus*, Clupeidae). The marine trophic index fluctuated at about 3.9 from 1950 to the mid-1980s with lower values at 3.8 for the last twenty years. Such a change can be explained by decreasing abundance and landings of cod, but also of other high trophic level species such as whiting and anglerfish.

(3) The West Scotland and Ireland ecosystem is the only one where the large fish indicator LFI exhibited a clear increase over the most recent period. This indicator was largely driven by the stock status of saithe, which collapsed in the early 1990 before recovering. This survey also moved into deeper water in more recent years, possibly introducing a bias towards larger fish. In contrast, other indicators based on surveys exhibited a consistent decline since the 1980s accelerating in recent years (mean maximum length MML from 90 to 70 cm, and the mean trophic level MTL from 4.0 to 3.4), while the marine trophic index only slightly decreased over the last decade (MTI from almost 4.2 to 4.0). This trend reflects the decreasing abundance of large fish predators such as cod and whiting. Indicators based on landings showed a decreasing trend over the past 60 years, with major changes occurring during the 1970s (the MTI went from 4.2 to 3.8, and the MML from 75 to 52 cm), when the blue whiting and mackerel fisheries developed, and a continuous but smaller decline over the 30 years, mainly explained by the decreasing abundance and catch of cod, and spurdog (*Squalus acanthias*, Squalidae).

(4) In the Irish Sea, length-based indicators from surveys (LFI and MML) exhibited a slightly increasing trend, but with absolute values smaller than in all other ecosystems (proportion of large fish LFI on average below 0.12, and mean maximum length MML around 42 cm). In contrast, trophic-based indicators sharply decreased over the past 20 years in the survey data and over the past 30 years in the landings (MTL from 3.5 to 2.8, and the MTI from 4.2 to 3.9). Changes observed in landings reflect the increasing catch of species like Norway lobster (*Nephrops norvegicus*, Nephropidae) but also the decreasing trend in abundance and catch of cod, whiting, saithe or hake (*Merluccius merluccius*, Gadidae). The mean maximum length MML from landings increased around 1980 mainly due to the collapse of herring.

(5) In the Celtic Sea, the large fish indicator LFI seemed to slightly increase over the last decade (in contrast to the results of Shephard et al. (2013), which were based only on the UK WCGFS survey). All other ecosystem indicators based on surveys remained stable over the study period (1993-2010) showing no clear sign of recovery that may be attributed to the observed decrease in fishing

pressure. This relative stability is put in perspective by complementary landings-based indicators suggesting that major changes already occurred in this ecosystem before the beginning of scientific surveys, with very strong decreases from 1950 to the late 1970s (the mean trophic level MTL went from 3.9 to 3.6, and the mean maximum length MML from 85 to 45 cm) and subsequently stabilising at low levels.

(6) In the Bay of Biscay, indicators were only available for a relatively short period (1997 to 2010). A slight increase was observed in length-based indicators from surveys (the LFI went from 0.09 to 0.15, and the MML from 52 to 56 cm), while trophic indicators remained stable. Time series of indicators based on landings were also shorter compared to other ecosystems. Since the mid-1980s, mean trophic levels appeared to be stable, while the marine trophic index and the mean maximum length of landed fish decreased around 2000, mainly driven by increased catches of horse mackerel.

(7) On the Iberian coast, available time series of surveys were even shorter (from 2002 to 2008) and hence it is difficult to draw conclusions on indicator trends. In contrast, indicators from landings were available since the 1950s. Mean trophic level and mean maximum length remained rather constant over the whole period, exhibiting the lowest values (about 3.4 and 43 cm respectively) of all ecosystems analysed in this study. The marine trophic index decreased (from 4.0 to 3.7) reflecting changes in landing composition, with hake landings in particular decreasing due to over-exploitation, and more blue whiting or mackerel caught over time.

In summary, despite of the difference in magnitude between ecosystems, a long term overall decline in the landings-based indicators was observed across all assessed European seas (Fig. 8). Thus, since 1950, the mean trophic level of landings has declined from 3.7 to 3.3, while the marine trophic index decreased on average from more than 4.0 to about 3.8. This trend is not modified when indicators are calculated using values of trophic levels from local Ecopath models, instead of the standard values from Fishbase (see detailed results of the sensitivity analyses in the Supplementary Material). Over the same period, the overall index of mean maximum length decreased from 68 cm to 49 cm. In other words, landings from European Seas progressively became dominated by smaller species and lower trophic levels.

Even if calculated over a shorter period, indicators based on surveys showed that the decrease in landings was not only related to putative changes in fishing strategy, but also to observed ecosystem change. The global trophic-based indicators confirmed a deterioration of the community structure of the ecosystems from 1985-2010 (MTL decreased from 4.05 to 3.80 and MTI dropped from almost 4.1 to less than 4.0). This decrease had an impact on all the 5 ecosystems where time series started before 1995, with the only exception being the Baltic Sea. Length-based indicators from surveys exhibited some similarities in their trends over the period. From the start of the time series both showed an initial strong decline (MML 90 cm to 74 cm, LFI 0.27 to 0.12) reaching a minimum in the early 1990s (MML) or early 2000s (LFI) after which the MML fluctuated below 80 cm while the LFI increased to 0.20 in 2010. This improving trend of the large fish indicator LFI over the last decade was clearly observed in the Bay of Biscay, but also to a lesser extent in the North Sea, the Irish Sea and the Celtic Sea.

It should be stressed that survey-based ecosystem indicators calculated for all ecosystems together are highly correlated with landing-based indicators ($r=0.77$, 0.92, and 0.80 for MML, MTL and MTI respectively). At the ecosystem level, a positive correlation was observed in 9 of 21 cases ($p<0.05$; see Table S4 in the Supplementary Material). Importantly, the recent declines in fishing mortality may only have resulted in an overall recovery of the fish community size structure while all other ecosystem indicators continue to decline.

Synthesis on trends over the last decade

The global picture of recent indicator trends (Fig. 9) highlighted several points:

- The decrease in mean fishing mortality rates was significant in the seven European ecosystems considered in the study. The same trend was observed for nominal fishing effort when data were available; in particular fishing effort in terms of kw*fishing day approximately halved between

2002 and 2010 for the North Sea, the Irish Sea, and the West of Scotland and Ireland (data from STECF 2012). Landings also decreased in all ecosystems.

- With the exception of the Baltic Sea, the decrease in the mean recruitment index was significant in all ecosystems.
- The spawning biomass of assessed stocks increased in several ecosystems, but remained at low levels (especially in the North Sea, the Irish Sea, and the West Scotland and Ireland), and is still decreasing on the Iberian coast. On the other hand, the large fish indicator (LFI) seemed to improve in several ecosystems, suggesting that the size structure of exploited stocks has started to recover. The observed decrease in recruitment may have counterbalanced the benefit expected from the release of the fishing pressure, leading to almost stable total biomass.
- Even when the total biomass of assessed stocks was increasing, several ecosystem indicators still declined suggesting ongoing degradation in ecosystem health. This suggests that the observed decrease in fishing pressure has not been sufficient or is still too recent to allow recovery of ecosystems from a depleted state, especially in terms of species composition and trophic biodiversity.
- Some contrasts do exist within ecosystems. In particular, based on the available indicators, the Bay of Biscay ecosystem seems in better shape, or showed stronger improvement than others. In contrast, many indicators exhibited deteriorating trends in the West of Scotland and Ireland ecosystem. However, due to the relatively limited availability of data in the Bay of Biscay and the Iberian Sea compared to the northern areas, this conclusion is cautious.
- More data and/or longer time series are available in the northern European seas (in the Baltic Sea, North Sea and West Scotland and Ireland). In particular, indicators based on stock-assessments (i.e. fishing mortality F, spawning stock biomass SSB, recruitment R and sustainable fishing mortality index F*) can be considered representative of the whole fished fraction of ecosystems. In contrast, most landings in other ecosystems are related to non-assessed stocks and are thus not included in some of our analyses due to data availability.

Discussion

Building ecosystems diagnoses in support of a science-based EAFM

. *Using landings-based indicators*

In order to implement an Ecosystem Approach to Fisheries Management (EAFM), an assessment of the status of marine ecosystems and of temporal change is required. In order to draw valid conclusions, the longest time-series available should be considered (Guénette and Gascuel 2012). In the European seas it emerged that surveys alone are not sufficient to build diagnoses on ecosystem health, as they only describe a relatively short period before which the system was already impacted and major changes had occurred. Catch or landing statistics are available over a longer period, but using such data to infer information about stock abundance or ecosystems health has been strongly debated among fisheries scientists (Branch et al. 2011; Carruthers et al. 2012; Pauly et al. 2013). Some of the observed changes since World War II reflect adaptations by the industry, either to ecological change (including that induced by fishing), consumer habits or markets. Changes also reflect developments in gear technology, which have allowed the emergence of new fisheries, for instance in deeper waters. Also, management and regulations as well as discarding practices have a significant impact on landings. Such changes may have a substantial impact on catch rates (Marchal et al 2006) and can thus cause bias in any landings-based indicator. In other words, some caution is required when interpreting catch or landing reconstructions and inferring changes over time since the latter may be influenced by both the species and the fleet segments included in the analysis (Essington et al. 2006; Thurstan and Roberts 2010; Heath and Speirs 2011). Nevertheless, landings reconstruction provides a long term perspective on exploitation history, which has to be kept in mind when attempting to assess ecosystem health in a more recent period of time. In European seas, landings showed that major changes took place from the 1950s to the 1970s, before contemporary scientific surveys started.

. *Using stock-based indicators and management targets*

Examining aggregated metrics based on formal stock assessment results was another important step in our approach towards an ecosystem approach to fisheries management. Using the results of single species stock assessments may not be perceived as the most obvious contribution to an ecosystem approach, but has also been recently applied to pelagic fish communities in the North Sea and Celtic Seas (Shephard et al. 2014). The current analysis shows that such an approach allows the compilation of stock-based indicators at the ecosystem level, using the best available estimates regarding the status of all the assessed stocks, and thus provides a useful diagnosis of state in the fished and assessed part of the ecosystem. The synthesis was based on F_{pa}, B_{pa} and F_{MSY} so that the status of each stock as well as their mean trajectories were defined with reference to both the “old” precautionary reference values and the new MSY-based reference value. According to the commitments of the 2002 Johannesburg world summit, the MSY-based objectives (implicitly defined based on the B_{MSY} target) should be reached, wherever possible, by 2015. The transition scheme, currently in force within ICES working groups, aims at the enforcement of this objective, but only considering F_{MSY} and not B_{MSY} as the new threshold (except for a few short lived species such as Norway pout), while maintaining B_{pa} as the SSB threshold even if B_{pa} is far below B_{MSY} for most stocks,.

In addition, the chance of the European Union to achieve the MSY objective by 2015 for all stocks seems highly unlikely (Froese and Proelß 2010; Villasante 2010). The Aichi Targets, defined at the Nagoya Convention on Biological Diversity (CBD 2010), set 2020 as the deadline to achieve MSY for all stocks worldwide, while the 2008 Marine Strategy Framework Directive (MSFD) implements the same 2020 deadline to achieve Good Environmental Status across all European waters, including for the MSFD Descriptor 3 which specifically addresses commercial fish and shellfish (European Commission 2008b and 2010). Thus, at present, both the MSFD and the newly revised Common Fishery Policy have adopted MSY as the reference level that should be reached by 2020 and applied this to the two indicators used to assess stock status: fishing mortality (F) and

Spawning Stock Biomass (SSB). Therefore, it will be especially interesting to monitor stocks trajectories (for each individual assessed stock or as a whole) in the coming years.

A limitation of our method is that reference points were not available for all stocks assessed by ICES, because they have not yet been estimated. The reference point $F_{0.1}$ derived from yield per recruit analyses could be used as a proxy of F_{MSY}, where no direct estimate of F_{MSY} is available (STECF, 2011). Nevertheless, this proxy is not often specified by ICES working groups, and thus in some instances only certain stocks could be considered in our calculations of mean stock trajectories. This was especially the case in the southern ecosystems we studied, where in general only a relatively small part of the catch comprised assessed stocks. However, an assessment based on the largest proportion of exploited resources should be considered an important requirement for achieving an ecosystem approach (including reference point estimates) and thus for the MSFD, specifically Descriptor 3 (on commercial fish and shellfish), but could also have relevance for the descriptors 1 (biodiversity), 4 (food web) and 6 (seafloor integrity). Such assessments would not necessarily be required on an annual basis and using the same full set of age-based methods. In particular for non-target species, where complete coverage is not realistic, a risk-based approach could be defined in order to assess key vulnerable species, and to determine the number of stocks necessary to provide a representative overall assessment of species exploited in each ecosystem.

There are, however, some issues to consider when attempting to apply these MSY-based reference points as part of an ecosystem approach because biological interactions may prevent achieving current single-species-based B_{MSY} thresholds simultaneously for all stocks (Piet and Rice 2004). Also since F_{MSY} is considered a limit reference value it may imply that some of the stocks caught in a multi-species fishery will need to be caught at levels below F_{MSY}. In addition, changing the size selectivity of the fishery will affect the values of the management threshold, and have consequence for ecosystem health leading to a smaller impact on marine resources (Brunel and Piet 2012, Froese et al. 2008). In other words, new targets will have to be defined, using multi-species and ecosystem models where the biological interactions (e.g. predation) and other ecosystem aspects of integrated stock sustainability are taken into consideration.

. *Using ecological indicators*

Ecological indicators are not routinely calculated for European ecosystems in any ICES or European working group or scientific program. The application of the MSFD indicators of Good Environmental Status to support integrated marine resource management is currently in a state of flux (Greenstreet et al. 2012; Lassen et al. 2013) which is nicely reflected by the indicators we considered which seem to capture relevant aspects of the fish community and probably also the wider ecosystem, and appear sensitive to the effects of fishing. In particular, trophic-based indicators appeared useful, highlighting a clear and continuous decreasing trend in several ecosystems as well as for the aggregated indicators. In addition, and in contrast to other studies (e.g. Branch et al. 2010 and 2012), trophic indicators based on landings appeared little sensitive to the uncertainty that exists regarding values of trophic levels per species. A decrease is also observed until the early 1990s for the mean maximum length indicators (MML). It reflects changes that have occurred in the species composition of demersal communities. In contrast, the LFI suggests the first signs of recovery from about 2000 onwards. If, indeed, the fishing pressure is decreasing as figure 4 shows, it suggests that the size structure of fish stocks is more sensitive than the species composition of fish communities during the start of the recovery phase.

Finally, this study showed that the landing-based indicators (MTL, MTI and MML) appeared to be highly correlated to survey-based indicators for most ecosystems, which contrasts with the results of Branch et al. (2010). Using a worldwide approach, these authors observed some negative correlations between landing-based and survey-based indicators and concluded that the mean trophic level from landings is an unreliable indicator of ecosystem health, potentially biased by changes occurring in fishing strategies. In European seas, we found that the decrease observed in the trophic indicators from landings may be partially due to changes which occurred in fishing strategies, with fishermen progressively targeting a wider part of ecosystems and landing more prey fishes. This reflects a ‘fishing through the food web’ process (Essington et al. 2006), which is confirmed by the observed increase in the index of landings diversity. However, the declining trend in mean trophic level was also observed in survey-based indicators, corresponding to a decrease in predator abundance. This reveals that a ‘fishing down the marine food web’ process has happened simultaneously, which also affects the mean trophic level of landings. Such a result reflects the global higher sensitivity to fishing of high trophic level species, due to their typically lower rates of turnover (Pauly et al. 1998; Gascuel et al. 2008 and 2011b). Analysing correlations between life-history traits and the occurrence of fish stock collapses, Pinski et al. (2011) showed that stocks of low trophic level species (e.g. small pelagics) may be more liable to collapse. This reflects their small number of age classes, but also frequently very high exploitation rates. In Europe, bottom trawl fisheries targeting large higher trophic level demersal species have been historically dominant. It is thus not surprising that many such species are currently very depleted, and that a fishing down process can be observed. More generally, the strong decrease in large demersal fish abundance and in the mean maximum length of survey data is a global pattern (Worm et al. 2009).

. Implementing an effective ecosystem approach to fisheries management

More research on ecosystem indicators is still needed and several research initiatives aimed at developing operational ecosystem indicators exist, such as IndiSeas (Shin et al. 2012), several ICES working groups or regional sea conventions such as OSPAR or HELCOM . Once the selection process of appropriate indicators begins to converge, the use of these indicators, as part of an ecosystem-based resources management towards the achievement of good environmental status, needs to be (further) developed and routinely enforced. It should be stressed that the objective of reaching the good environmental status is required but will not be sufficient. As such ecosystem approach to fisheries management goes beyond ‘just’ ensuring Good Environmental Status. It aims to take into account not only ecological sustainability, but also economic profitability and social fairness (Garcia and Cochran 2005; Gascuel et al. 2011a; Bundy et al. 2012). In other words, its major objective (its specific value-added) is to analyse trade-offs between ecology, economy and social aspects, the three pillars of the sustainable development of fisheries (Gascuel et al. 2012; STECF 2010). In addition, according to several European directives where responsibility has been delegated by Member States to the EU, fisheries management in European Seas is an integrated policy. Therefore, the ecosystem approach to fisheries management could (or should) be implemented at the European level, while environmental policy and therefore enforcement of the MSFD have to be conducted at the national level.

Finally, this study shows that the large ecosystems considered in the present analyses, according to the reference list defined by STECF (2012), represent a good compromise in terms of size and the appropriate scale to synthesise stock status and analyse trends in the ecosystem indicators and can be easily aligned to MSFD subregions. These ecosystems are similar to eco-regions used by ICES except that two of the large ICES eco-regions (i.e. 'Celtic Sea and West of Scotland', and the 'Bay of Biscay and Iberian Seas') have each been sub-divided into two ecosystems. The availability of the data we used, as well as the results we obtained, seem to validate these four subsystems, with notably contrasted trends and diagnoses from one ecosystem to the other. The seven ecosystems we considered also appear to be appropriate for the study of ecological impacts and economic performances of fleet segments, and to analyse trade-offs between economy and ecology in order to develop fleet-based management of fisheries in the frame of an operational ecosystem approach to fisheries management (STECF 2010 and 2012; Gascuel et al. 2012). They also should be the basis to develop ecosystem models devoted to scientific advice on both ecology and economics, and to define long term management plans in support of the Common Fisheries Policy. Finally, they form "territories" where dialogue should be improved and stakeholders involved in participative management of fisheries, in line with the strengthening of the role of the Regional Advisory Councils (RACs) established by the European Commission. Therefore such ecosystems should be the basis for the collection and availability of data required for the further development and application of the ecosystem approach, which is still a major concern in areas including the Mediterranean and Black Sea (Coll et al. 2013).

A global diagnosis on fishing impact in European seas

European seas have been exploited for millennia (Lotze et al. 2006) and fishing has been identified as a major human impact in these systems (Narayanaswamy et al. 2013). Motorised boats as well as large bottom trawls have been used since the end of the 19th century (Herubel 1912) but until World War II, catches remained relatively low. Only about 1.4 million tonnes were landed in 1938 from the North Sea, by far the most productive area within European Seas (Christensen et al. 2003). Trends in fishing effort and fishing mortality rates strongly increased from the 1950s until the 1990s. For instance, the overall French nominal fishing effort increased from 164 thousand KW in 1950 to 500 thousand in 1970, and to 750 thousand in the late 1990s (Guénette and Gascuel 2012). In the North Sea, fishing mortality of herring (the most important species in term of landings) jumped from 0.2 in 1950 to more than 0.9 in 1970, while it increased for cod from 0.48 in 1963 (the first available year in assessments), to 0.8 in the mid-1970s and to around 1.0 during the 1980s and 1990s (ICES, 2013). In the North Sea, where some ICES assessments started early, overexploitation had occurred by 1953 for herring and sole, 1957 for plaice, 1963 for cod, haddock and whiting, and 1967 for saithe (ibid.). Due to limitations in data availability at the ecosystem scale, our analyses did not provide results on trends in stock abundance before the 1980s, but several studies show that fish stocks had already declined in Europe before that period. Using a CPUE index for the Celtic Sea and the Bay of Biscay, Guénette and Gascuel (2012) estimated that the overall abundance of exploited species declined by around 80% between 1950 and the late 1970s. Based on stock assessment results for the 1970s, a very strong decrease in the mean spawning biomass was also recorded by Garcia and De Leiva Moreno (2005) for a selection of 14 large European stocks, and by Froese and Proelß (2010) for 54 fish stocks of the Northeast Atlantic.

Until the 1970s, total landings increased reaching more than 7.2 million tonnes for the seven considered ecosystems, of which 3.4 million tonnes came from the North Sea. It should be noted that the data available for this period are likely to underestimate total catches due to discarding practices and unreported landings (e.g., Pitcher et al. 2002, Zeller et al. 2011). The number of species exploited at that time was relatively low, with fishermen mainly targeting the most abundant stocks, often consisting of large predatory species (especially cod, but also whiting, saithe, hake and tunas). In the North Sea, an increase in gadoid recruitment occurred in the late 1960s (the 'gadoid outburst', Cushing 1980), leading to an increase in biomass and catch. This increase occurred as the various herring stocks were in decline, probably under the stress of heavy fishing (ibid.). It temporarily masked or postponed the decline of fisheries for herring and other species, which may have commenced by the late 1960s.

From the mid-1970s yields began to decrease everywhere (the Baltic Sea being an exception) as a result of the overexploitation of some major stocks. In order to compensate for their losses, fishermen adapted by increasing fishing effort and targeting a wider range of species, and thus exploiting a wider part of ecosystems. In particular small pelagic species, but also invertebrates, and since the late 1980s deep-water species, have increasingly been targeted. Thus, the diversity of landings increased, and this trend persisted until the late 1990s. Nevertheless, this was insufficient to compensate for the decrease in abundance of many stocks, and total landings have continuously declined in all the NE Atlantic ecosystems studied. Both the landing-based and survey-based indicators revealed a trend of increasingly deteriorating ecosystems, at least until the late 1990s. It is likely that significant fishing-induced changes in the species assemblages have occurred, with increasingly lower abundances of vulnerable species, especially large predators. According to the 'fishing down marine food web' process, higher trophic levels were the most affected by overfishing, while prey species (at least some of them) might have benefited from the release of predation. This release from predation pressure, but also the impact of trawlers on the seafloor, may partly explain, in some areas such as the west of Scotland, the dramatic shift from whitefish dominated fisheries to fisheries dominated by Norway lobster and scallops (Thurstan and Roberts 2010).

The highest fishing impact on the environmental status of European seas seems to have taken place in the late 1990s, with high fishing mortality rates and a clear decrease in stock-based and ecosystem indicators. In 1998, the EU formally adopted the ‘precautionary approach’, using B_{pa} and F_{pa} as threshold to calculate total allowable catch and quotas. Scientific advice based on those thresholds was not always followed. Piet et al (2010) showed that for only 8% of the 125 stocks for which ICES provided advice over the period 1987–2006 the official total quotas equalled scientific advice, while the official quotas overshot scientific advice by >50%. Nevertheless, in the short term this approach led to more restrictive quotas and may have contributed to the decrease in fishing mortality rates observed in the following years. This decrease is expected to continue now that the more precautionary thresholds based on MSY are adopted and the uptake of scientific advice appears to have improved. In addition, other restrictions like fishing gear limitations or fishing effort quotas were enforced at that time as part of the implementation of long term management plans. Decommissioning schemes within the EU have also had some success in reducing fleet capacity. However the level of reductions varies between ecosystems, and the rate of reduction has been criticized as being too slow to counter other technological advances (European Commission 2012 and 2013). It should also be stressed that B_{MSY} is still not used as a target for fisheries management in Europe, assuming that as long as the reproductive capacity of stock is not compromised (i.e. $B \geq B_{pa}$) and fishing mortality is at a sustainable level (i.e. $F \leq F_{MSY}$) stocks should ultimately recover to their ecosystem-based MSY level. However, B_{pa} is the edge of safe biological limits and a stock at that size can be considered as probably safe but not as in good condition.

Overall, the mean fishing mortality rate of assessed stocks has almost halved over the last decade. This may have contributed to the accelerated decrease in landings observed in the recent years, notably in the North Sea. In spite of this dramatic decrease in fishing pressure, a large number of stocks still failed to meet the requirement of the 'past' B_{pa} or F_{pa} targets (11 stocks among 21, in our analysis based on 2013 assessments). No clear recovery in the biomass is currently apparent and the total biomass remains close to B_{pa}, a level far below B_{MSY}. Several recent meta-analyses (Cardinale et al. 2013, Fernandes and Cook 2013), and oral presentations made by ICES experts under the auspice of the European commission (Fernandéz, 2013), confirmed the strong reduction in mean fishing mortality of NE Atlantic fish stocks over the last decade. Nevertheless, these analyses also suggested a significant increase in the mean biomass of these stocks, while we observed no clear positive trend. In fact, these apparent contradictions are linked to differences in methods. On the one hand, we only considered stocks in European seas, while the ICES conclusions were based on stocks of the whole Northeast Atlantic, including Norwegian waters and Barents sea where several large stocks are currently recovering very rapidly (especially the Barents sea cod stock). On the other hand, ICES used as an index the average of standardised biomasses, while we considered the total biomass. Thus, results indicate that several small stocks do exhibit positive trends in biomass, while some large stocks would still be in decline.

Regarding ecosystem indicators, the large fish indicator LFI is the only one showing a positive trend, suggesting that the age and size structure has started to recover for some exploited stocks. But at the same time, other indicators of community structure (MML, MTL and MTI) continued to deteriorate or remained at low levels. This result suggests that the decrease in the fishing pressure has not yet been sufficient, or is still too recent, to allow for the recovery of ecosystem health from such a depleted state. Synergic effects between stock size, age structure, recruitment, ecosystem indicators, may explain the observed delay or lack of rebuilding. Other factors, especially related to environmental and global change, might also intervene and counterbalance or delay the benefit expected from the release of the fishing pressure.

In this context, the observed decrease in our overall recruitment index was an unexpected result which requires particular attention. In some cases, where fishing patterns change over time, recruitment estimates based on cohort analyses can be biased (e.g. Fonteneau et al., 1998).

However, it is very unlikely that such biases would affect all or even most of the ICES assessments and the decrease in recruitment appeared as a consistent pattern observed over a long period (more than twenty years) in all the European ecosystems we considered across many stocks (except for the Baltic Sea). Many stocks were close or below B_{pa} for a long time and their age structure is such that experienced, large, fecund females with high-quality eggs have disappeared due to overfishing and fishing of juveniles. Thus, recruitment overfishing may explain, at least partially, the decrease in recruitment. Several ecological mechanisms, potentially synergistic, should also be considered as additional hypotheses able to explain this trend. In particular: the impact of climate change on ecosystems productivity and food web; anthropogenic impacts on essential habitats (especially on coastal nursery grounds); fishing impact on seafloor and benthic productivity (Hiddink et al. 2006); unexpected ecosystem effects due to changes in species assemblages; and the loss in the genetic biodiversity for some severely overexploited species (Heath et al. 2013). In all cases, such a decrease in the mean recruitment index for a large set of assessed European stocks is of great concern for fisheries and may explain, at least partially, why the biomass of exploited stocks has not yet recovered, despite declining fishing pressure and possible recovery in the length-based LFI. In the coming years, poor recruitment may lead to reduced catches and higher fishing mortalities (for a given quota) than expected from stock assessments, therefore compromising the effectiveness and the social acceptance of fisheries management. It would be difficult to follow the transition scheme and to reach the new MSY target in such a context, and thus to achieve Good Environmental Status under the EU MSFD by 2020. In the medium term, if the trend continues, it could also lead to a decrease in absolute landings, with obvious potential impacts on fisheries profitability. In addition, it should be stressed that in many instances, targets set for ecological indicators are established on the basis of historical indicator values. If physical and hydrographical conditions are now very different from those prevailing in the early 1980s, such that recruitment indices have all declined markedly right across European waters, then the validity of such targets as representing the good environmental status of ecosystems must surely be called into question.

Finally, the consistency of diagnoses that we established in the various European Seas has to be underlined. Trends and global status appeared quite similar and only small contrasts have been identified between ecosystems, with for instance more favourable trends in the recent period in the Bay of Biscay compared to the West of Scotland and Ireland. The Baltic Sea is an exception, with several indicators exhibiting contrasting trends compared to other ecosystems. In this ecosystem, total landings only slightly decreased over the last decade, and the mean recruitment index fluctuated with no clear trend over the whole period. The specifics of the Baltic Sea can at least partly be explained by the synergic effect of a low number of main fish species in the ecosystem (only 3), and low predation rate due to unfavourable reproduction conditions of cod, effectively contributing to the high numbers of sprat stock since 1990s. Increased eutrophication can also have a role in elevated numbers at lower trophic levels.

Conclusion

The working group on the Ecosystem Approach to Fisheries Management (STECF 2012) noted three key element that constitute the work that has to be performed on a regular basis to evaluate and implement a scientific-based ecosystem approach to fisheries management in European Seas (and probably everywhere):

(i) Diagnoses of ecosystem health such as the one we present in this study have to be defined and regularly updated for each ecosystem, in close cooperation with MSFD implementation;

(ii) Both environmental impacts and socio-economic performance of the various fleets operating within each ecosystem have to be assessed and monitored. Results of such analyses could and probably should be considered by stakeholders (including the European Commission) in the definition of management options and especially in the frame of long term management plans (which should evolve from a stock-based to an ecosystem-based approach);

(iii) One or a limited set of ecosystem and bio-economic models should be set up and used on a regular basis for advice-oriented purposes. In a manner similar to the way assessment and forecast models are used for stock-based advice and management (and quotas or fishing effort regulation), ecosystem and bio-economic models should be regularly updated. This would constitute a key step in assessing the ecosystem impacts of fisheries, to simulate various management options and to analyse their potential effects on fisheries social-economic performance as well as ecosystem impacts.

Overall, it can be concluded that building ecosystem state diagnoses is a key first step on the path to evaluating and implementing an EAFM. We presented here the first diagnosis for seven of fourteen European marine ecosystems that have to be considered in the context of EAFM implementation in European seas. The present work should be considered as a starting point for more complete approaches, covering more ecosystems and the implementation of subsequent monitoring programs.

Acknowledgements

This study was initially conducted during the 2012 meeting of the Experts working group on the “Development of the Ecosystem Approach to Fisheries Management in European Seas” set up by the Scientific Technical and Economic Committee for Fisheries (STECF), and funded by the European Commission. MC was funded by the European Commission through the Marie Curie CIG grant and the Spanish Research Program Ramon y Cajal. We sincerely thank Trevor Branch and anonymous referees for their useful comments and suggestions on an earlier version of the manuscript.

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Figure captions

Figure 1 – Study area and boundaries of the seven marine ecosystems considered as case studies, according to STECF, 2012.

Figure 2 – Trends in annual landings (thousand tonnes) in 1950-2010 from European (in light grey) vs. non-European (dark grey) countries and catch time series (black line) derived from stock assessments in the seven marine ecosystems and for all ecosystems considered together.

Figure 3 – Trends in the indices of landings diversity. A and B: number of exploited species (for which landings are higher than 0.5% of the mean 2001-2010 yearly total landing). C and D: Shannon’s diversity index (H).

Figure 4 - Trends in stock-based indicators: mean fishing mortality F (column A, in year^{-1}), total spawning stock biomass SSB (column B, in thousand tonnes), and the mean recruitment index R (column C, relative value to the 1990-2000 average), for all the 57 stocks assessed by ICES in European seas (first line) and by ecosystem. (The red line refers to all stocks assessed in 2012, while the blue line is the longest available time series including at least 60% of assessed stocks)

Figure 5 - Current status of all assessed stocks, in relation to fisheries advice and management targets B_{pa}, F_{pa} and F_{MSY}. Note that only 21 stocks having associated target limits were considered here. On these graphs, the horizontal line labelled ‘B_{pa}’ refers to B* equal to 1 ($B^*=B_{current}/B_{pa}$), while the vertical lines labelled ‘F_{MSY}‘ and ‘F_{pa}’ refer to F* equal to 0 and 1, respectively ($F^*=(F_{current}-F_{MSY})/(F_{pa}-F_{MSY})$). The white sector relates to situations where the ‘new’ F_{MSY} management targets are met, the dark grey indicates stocks that do not follow the former ‘precautionary approach’, and light grey indicates stocks falling between the ‘old’ F_{pa} and the ‘new’ F_{MSY} targets.

Figure 6 – Mean temporal trajectories of assessed stocks within each of the studied ecosystems and for all European seas. The white sector relates to situations where the F_{MSY} management targets are met, the dark grey indicates stocks that do not follow the former ‘precautionary approach’, and light grey indicates stocks falling between F_{MSY} and F_{pa} targets. Only stocks for which target limits are known are considered (the Bay of Biscay and Iberian coast are not displayed because targets were only known for one stock); the percentage indicated for each ecosystem refers to the part of the ecosystem landings due to stocks considered on the graph.

Figure 7 – Trends in ecosystem indicators in the seven marine ecosystems. Column A: length-based indicators from surveys; column B: trophic level-based indicators from surveys; column C: indicators from landings. LFI = large fish indicator (proportion); MML = mean maximum length (cm, axis on the right); MTL = mean trophic level; MTI = marine trophic index.

Figure 8 – Mean trends in ecosystem indicators: a. length-based indicators from surveys, b. trophic level-based indicators from surveys, c. indicators from commercial fishery landings. LFI = large fish indicator (proportion); MML = mean maximum length (cm); MTL = mean trophic level, MTI = marine trophic index (Dot lines in graph c relate to the sensitivity analysis, using trophic levels from local Ecopath models in place of standard values from Fisbase; see Supplementary Material)

Figure 9 – Summary of trends over the last 10 years in the main indicators of ecosystem health in the seven ecosystems considered: total landings Y, fishing effort E, mean fishing mortality rate F, total stock spawning biomass SSB, mean recruitment index R, index of mean sustainable fishing mortality F*, survey large fish indicator LFI, mean maximum length MML from surveys or from landings, mean trophic level MTL from surveys or from landings, % of landings due to assessed stocks. Green and red symbols refer to positive and negative trends respectively (i.e. improving or deteriorating stocks status), while black arrows refer to uninterpretable changes in trend (landings might for instance decrease either because F or B decreases)

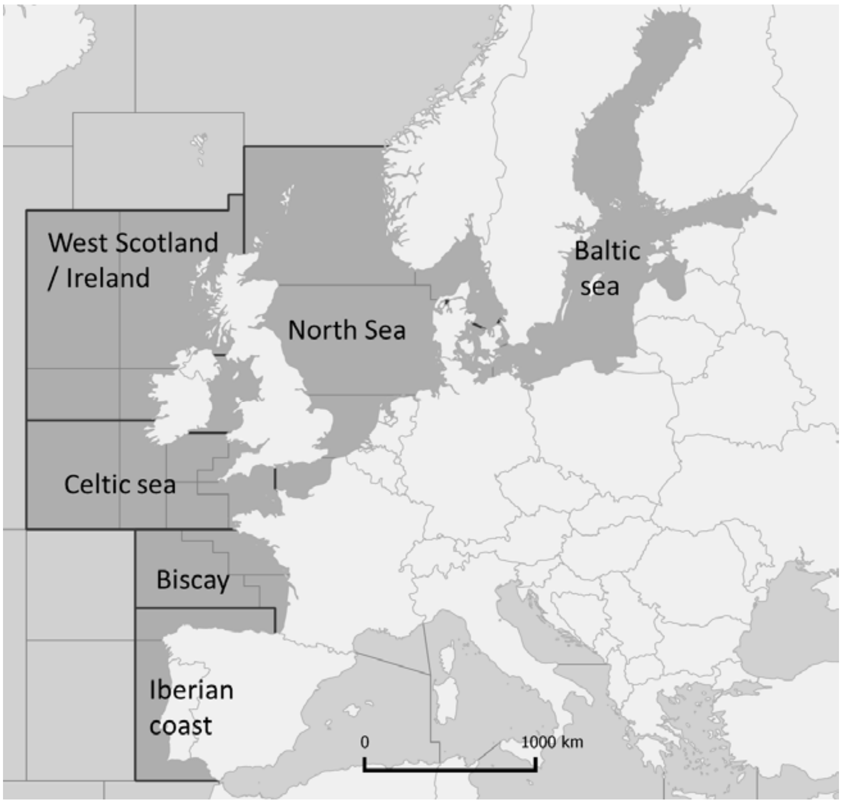

West Scotland
/ Ireland
North Sea
Baltic
sea
Celtic sea
Biscay
Iberian
coast
0
1000 km

Figure 1

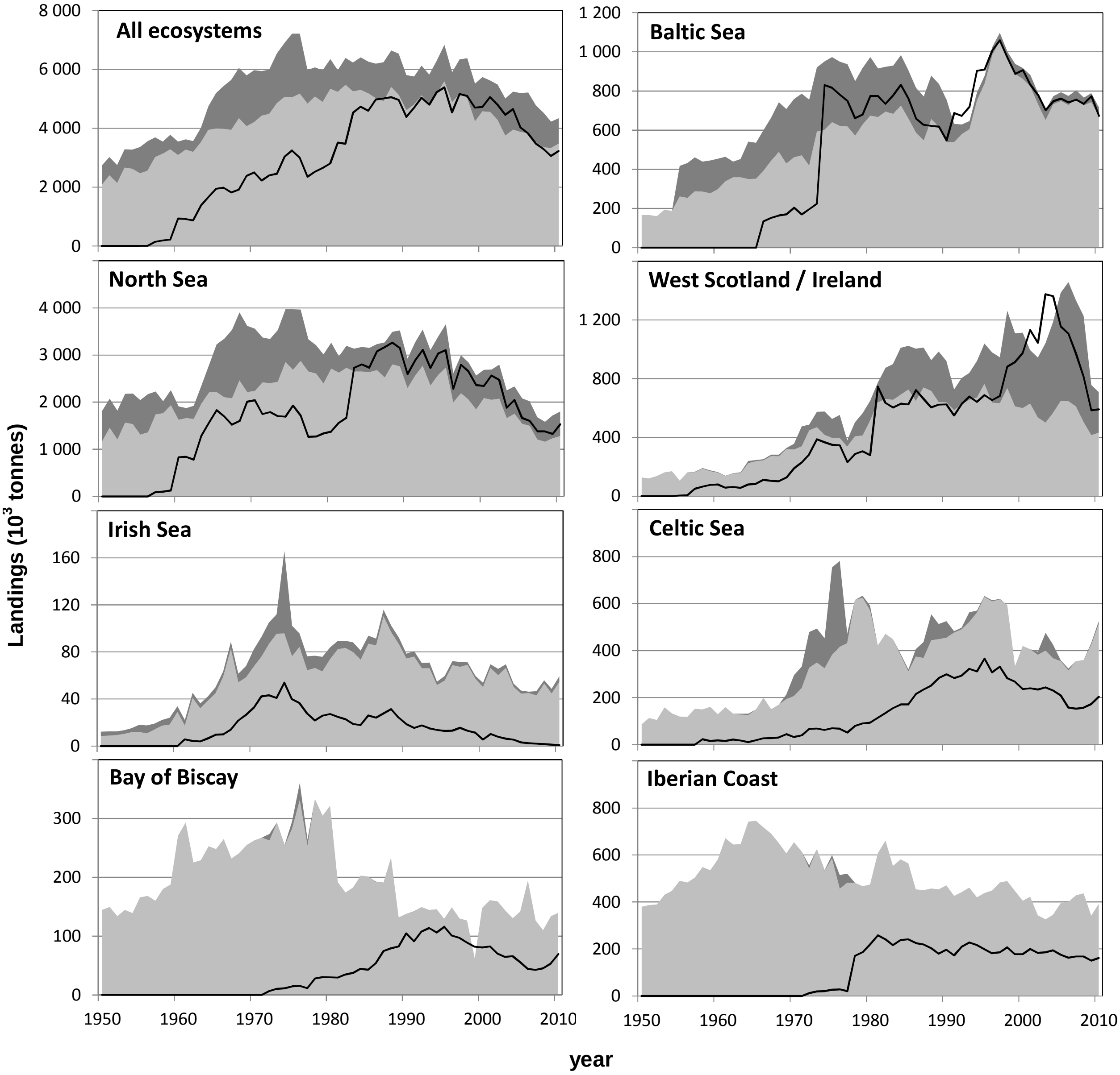

All ecosystems
Baltic Sea
North Sea
West Scotland / Ireland
Irish Sea
Celtic Sea
Bay of Biscay
Iberian Coast
Landings (10^3 tonnes)
year
8 000
6 000
4 000
2 000
0
1 200
1 000
800
600
400
200
3 000
1 000
160
120
80
40
300
100
1950
1960
1970
1980
1990
2000
2010

Figure 2

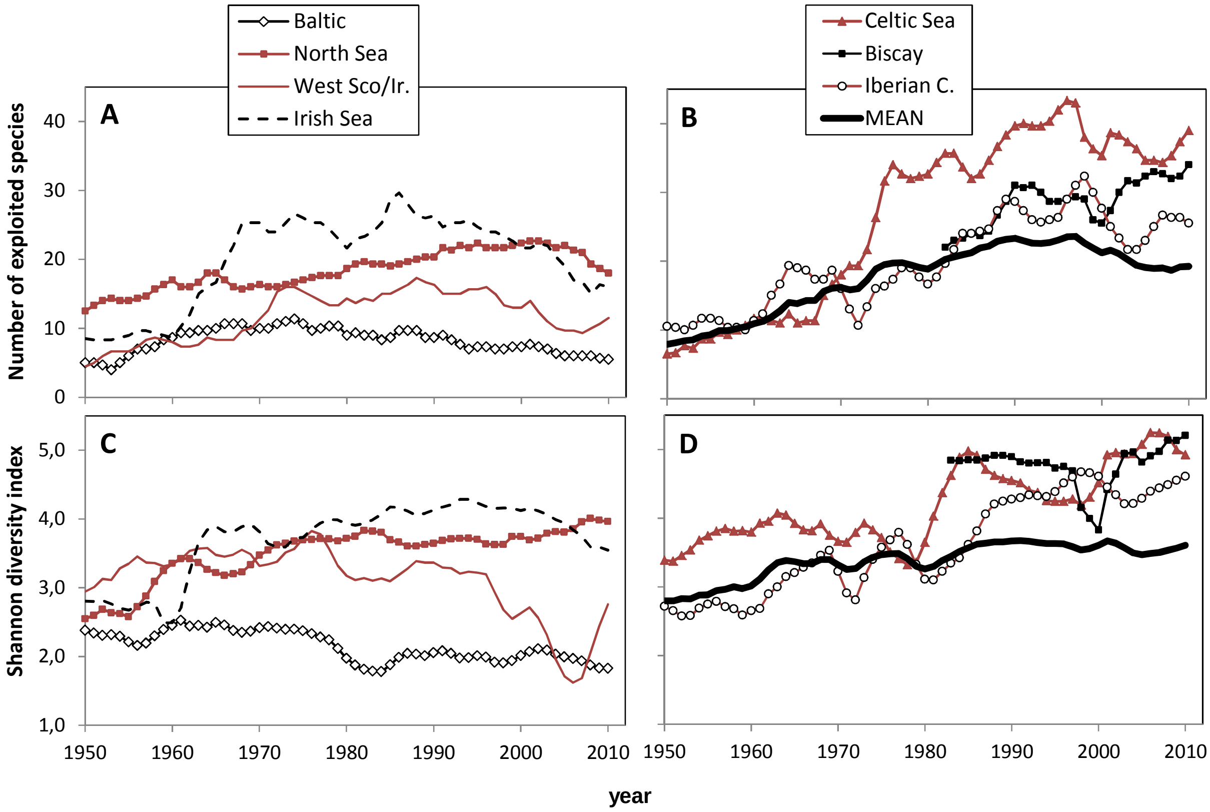
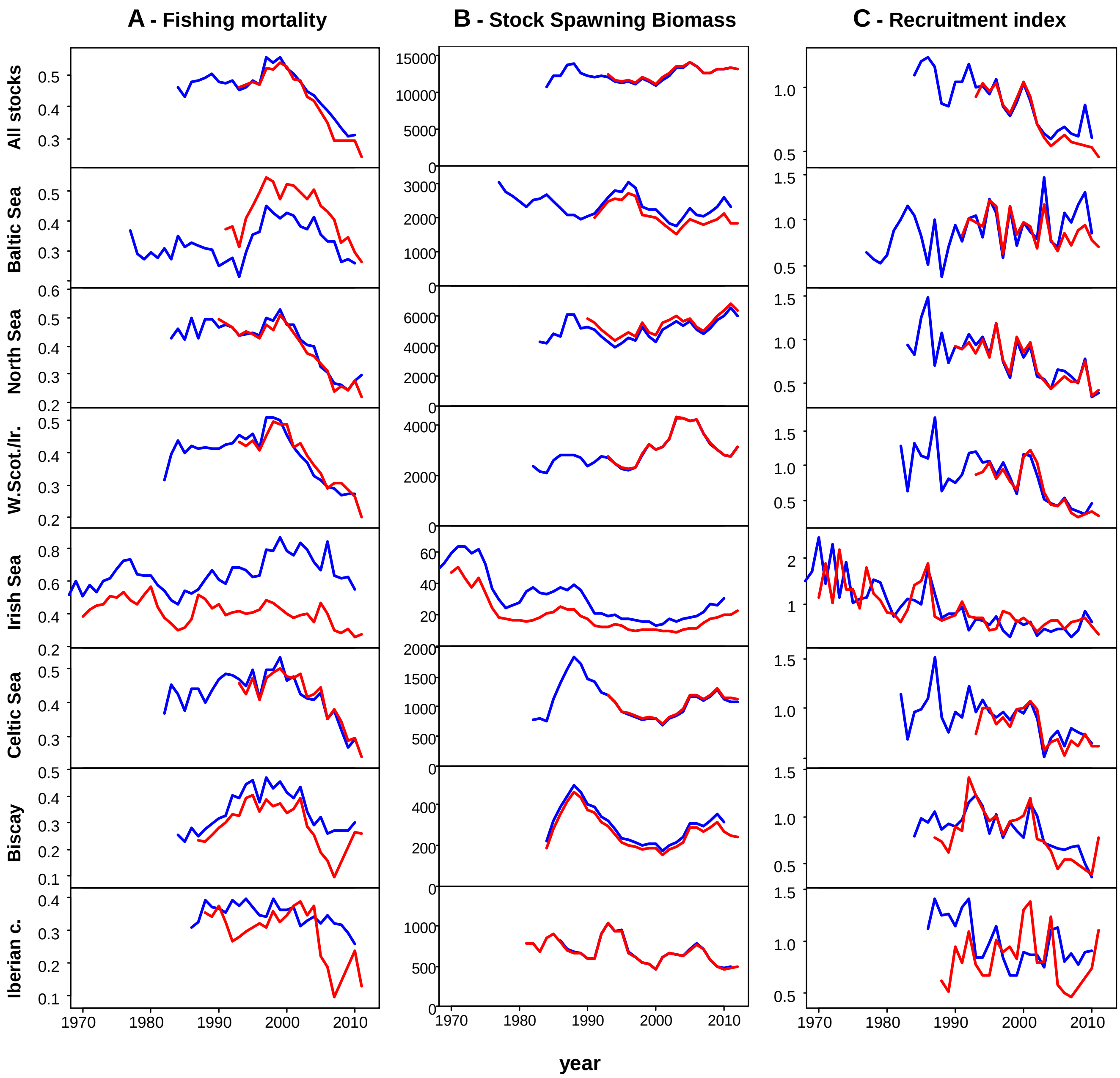

Baltic
North Sea
West Sco/Ir.
Irish Sea
Celtic Sea
Biscay
Iberian C.
MEAN
A
B
C
D
Number of exploited species
Shannon diversity index
0
10
20
30
40
1,0
2,0
3,0
4,0
5,0
1950
1960
1970
1980
1990
2000
2010
year

Figure 3

1121



1122

1123 Figure 4

1124

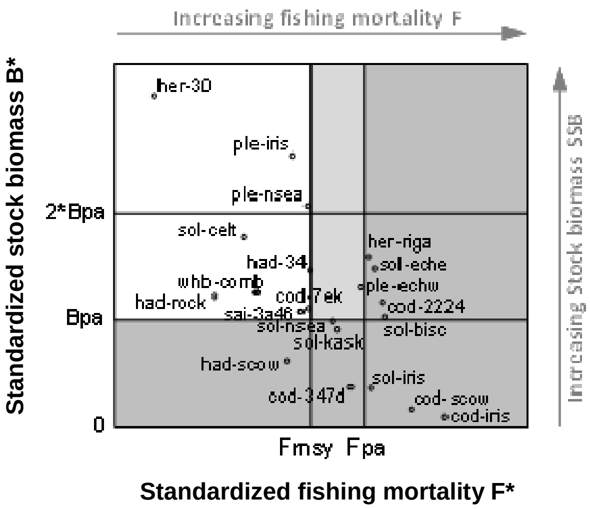


Figure 5

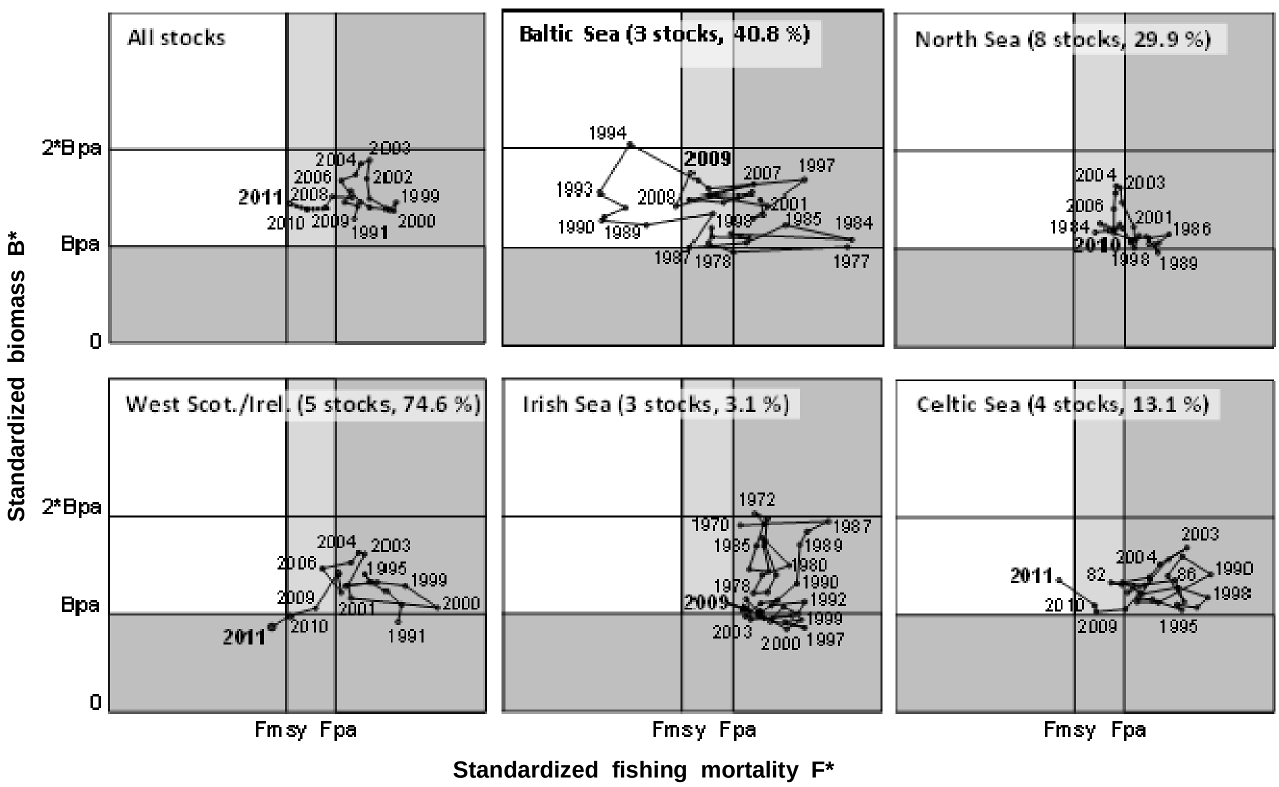


Figure 6

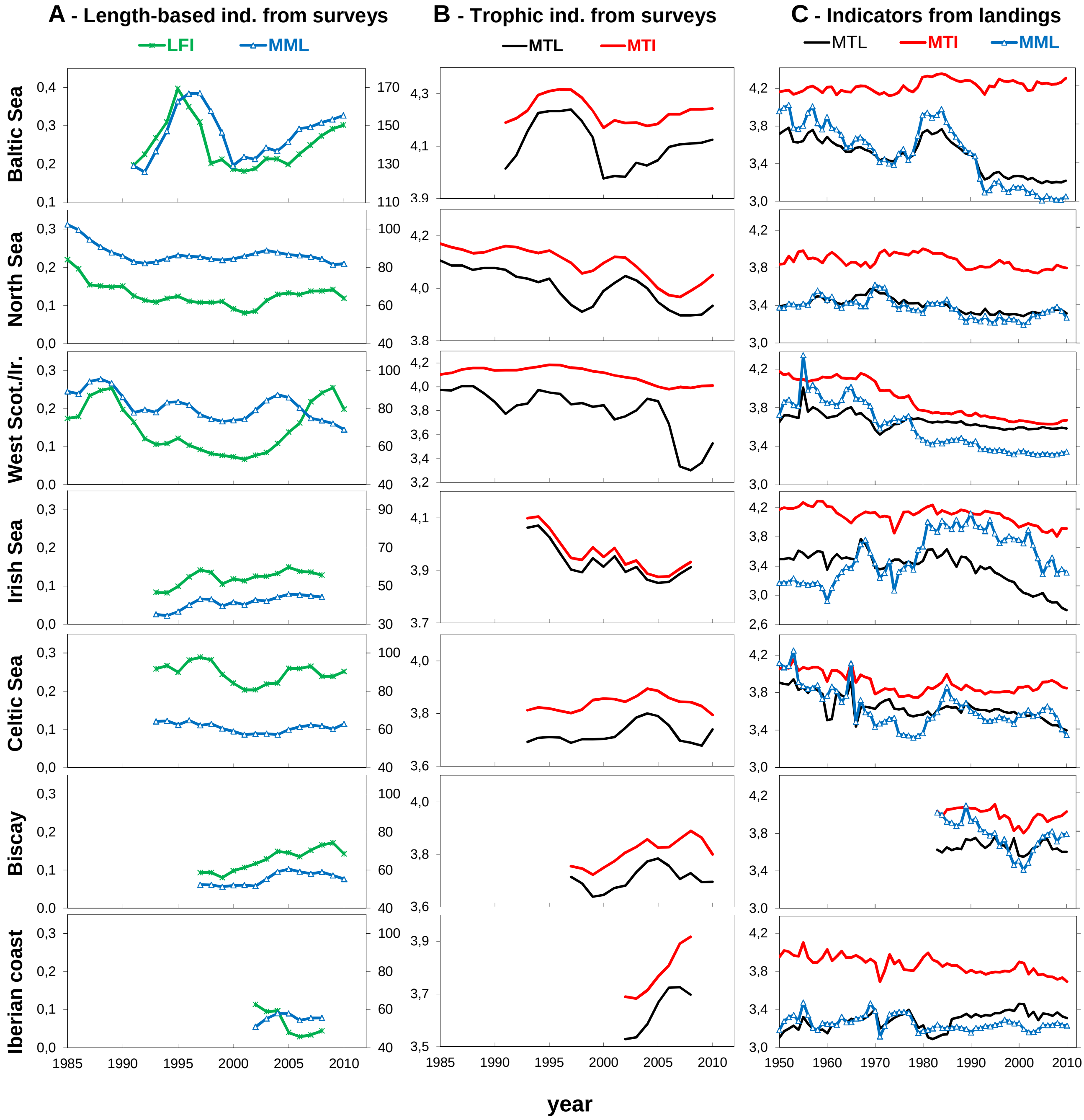

A - Length-based ind. from surveys
LFI
MML
B - Trophic ind. from surveys
MTL
MTI
C - Indicators from landings
MTL
MTI
MML
Baltic Sea
North Sea
West Scot./Ir.
Irish Sea
Celtic Sea
Biscay
Iberian coast
year

Figure 7 –

.

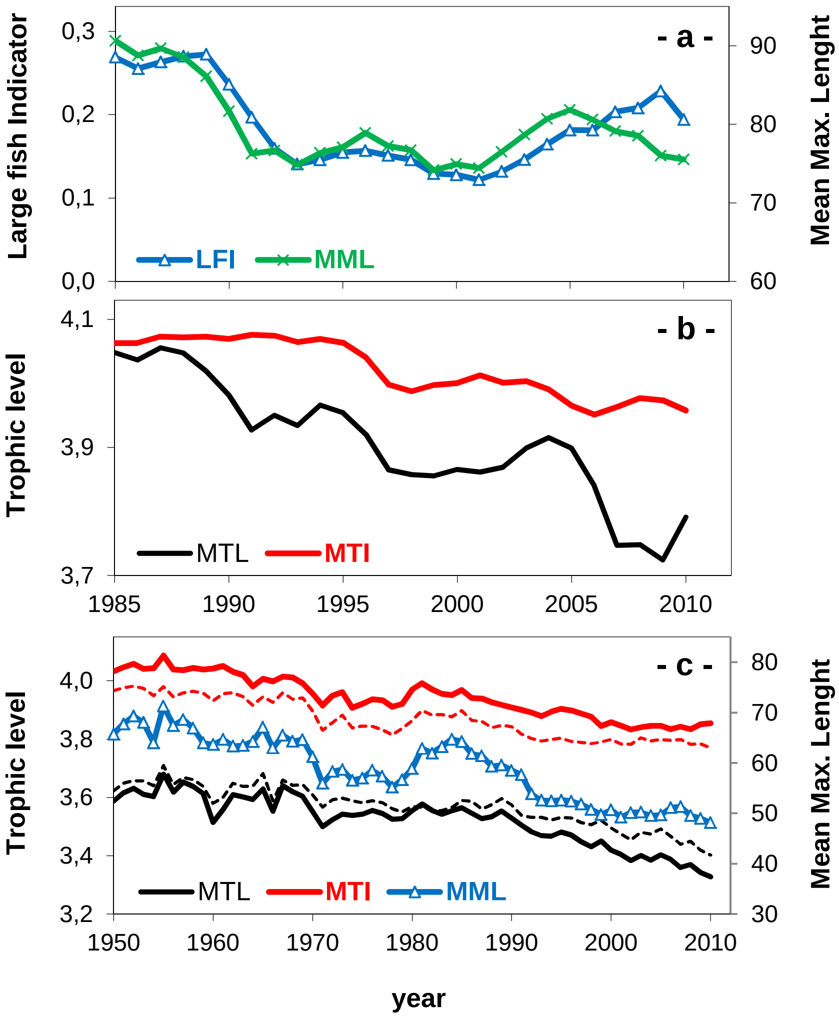


Figure 8

		Land. Y	Effort E	Mortal. F	Biom. SSB	Recr. R	Sust. F* B*	Survey LFI	Survey MML	Survey MTL	Land. MML	Land. MTL	% asses.
Baltic Sea		↘		↘	?	→	😐	↗	↗	↗	↘	↘	≈ 95
North Sea		↘	↘	↘	↗	↘	😐	low	↘	↘	↗	↗	≈ 85
North western Atlantic waters	West Scot./Irl.	↘	↘	↘	?	↘	😐	↗	↘	↘	low	low	≈ 90
	Irish Sea	↘	↘	↘	↗	↘	😐	low	↗	?	↘	↘	≈ 35
	Celtic Sea	↘	↘	↘	↗	↘	☺	→	↗	↘	↘	↘	≈ 40
South western Atlantic waters	Bay of Biscay	↘		↘	↗	↘	?	↗	→	?	↗	→	≈ 45
	Iberian Coast	→		↘	↘	↘	?	↘	→	↗	↗	↘	≈ 40

Figure 9