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## Shifting baselines in European fisheries: The case of the Celtic Sea and Bay of Biscay

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### ABSTRACT

Stocks status and ecosystem indicators of the Celtic Sea and the Bay of Biscay were analysed from 1950 to 2008 based on landings, stocks assessments data and additional auxiliary observations. The apparent stability in landings over the last 50 years masks the fact that the fisheries have been sustained at the cost of a dramatic increase in fishing pressure, and a change in species composition and fishing grounds. Major changes occurred between 1950 and 1970s with a major decrease in the global biomass index. The mean asymptotic length of fish landed has declined by 32 cm (19 cm for demersal fish alone) and the mean trophic level by 0.25. Both the Primary Production Required by the fishery and the Fishing in balance index have declined since the 1980s. All indices lead to conclude to a pervasive overexploitation over the last 30 years. Most exploited species considered are characterized by a severe truncation of their length and age structures, the reliance of the fishery on new recruits, and a large proportion of immature individuals in the landings. For most assessed stocks, fishing mortality is higher than  $F_{max}$  and close to  $F_{pa}$ . Rebuilding the stocks will require a 2–3-fold decrease in fishing mortality.

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### 1. Introduction

Fishing has been documented worldwide to decrease population abundance, truncate age and length structure and modify species composition. High trophic level predators in the North Atlantic have declined in abundance by two-thirds since the 1950s (Christensen et al., 2003) and only 59% of the stocks show a decrease in the rate of decline (Hutchings et al., 2010). More generally, the stocks sizes of demersal fish species have currently been divided by 5, or even by 10 or more, by overfishing (Froese and Proelß, 2010; Froese et al., 2008; Worm et al., 2009). Although fishing began more recently in West Africa, both landings and biomass declined since the 1970s (Christensen et al., 2004; Gascuel et al., 2007). In European Union waters, the decline of several target stocks has been halted, but the situation continues to deteriorate for others (ICES, 2010b; Sparholt et al., 2007; Villasante, 2010). For instance, 80% of EU stocks assessed by the International Council for the Exploration of the Sea (ICES) are fished beyond levels to the

maximum sustainable yield (European Commission, 2009; ICES, 2009).

The Celtic Sea and Bay of Biscay ecosystem has been fished for more than a century and is currently one of the major fishing zones for the European Union in terms of landings (Gascuel et al., 2010). After the World War II, the fisheries resumed with the support of governmental subsidies to modernize the fleets. Until the 1990s, the industry received subsidies to build new boats from national and European institutions, resulting in an unprecedented increase in fishing capacity in the region (Mesnil, 2008).

Pinnegar et al. (2002) have shown that the trophic level has declined in both landings and survey data between 1948 and 1998 in ICES divisions VIII–j (a subset of the present study area). Rochet et al. (2005) used survey data and seven derived indicators to assess the ecosystem status and trends for the Bay of Biscay and the Celtic Sea starting in 1987 and 1997 respectively. Both ecosystems were considered already impacted by fisheries in the initial year and the study concludes that they did not improve and even deteriorated (Celtic Sea). Nevertheless, the various population and community indicators showed inconsistencies in trend direction, possibly caused by variations in environmental factors, punctual recruitment events, and variance in surveys, resulting in less detection powers.

Reconstructions of historical landings have often brought forward the long forgotten past state of exploited populations and

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the extent of the loss incurred. This loss of memory of historical data causes the reference baseline of resources abundances to change with each generation of scientists and fishermen, a phenomenon that has been coined “shifting baselines” (Pauly, 1995). The present paper aims to 1. go back in time to gain a long-term perspective about landings and population abundance of the Bay of Biscay and Celtic Sea; 2. assess the long-term effect intense fishing on populations and the ecosystem; and 3. assess the current status of the related stocks and fisheries, in light of this long-term perspective.

## 2. Methods

### 2.1. Study area and data sources

The study area is defined as the continental shelf of ICES divisions VIIe–k and VIIIabd, known as the Celtic Sea and the Bay of Biscay respectively (Fig. 1). This delimitation of the study area corresponds to the spatial distribution of several stocks of exploited species such as cod, and the future use of this data for ecosystem modelling. Major fishing countries in the study area are France, Spain, and United Kingdom while Belgium, Ireland, and Germany also report landings.

Total landings for the study area were compiled for the period 1950–2008 and since 1914 for cod (*Gadus morhua*) and hake (*Merluccius merluccius*). Landings were obtained from FAO data sets before 1973, ICES Statland database (<http://www.ices.dk/fish/statland.asp>) completed/corrected with landing estimates from the ICES Working Groups for the assessed species, and ICCAT statistics (ICCAT, 2008) for albacore (*Thunnus alalunga*). This species is mainly caught at the break of the shelf of the Bay of Biscay and

offshore. Hake and cod landing statistics before 1950 were taken from the printed “Bulletin Statistique des Pêches Maritimes des pays du nord de l’Europe” published by ICES.

The main species group includes fish species for which stock assessments have been carried out and the main crustaceans: lobsters and crabs (Table 1). Other species were grouped in functional groups based on their size and the type of habitat they use (demersal, pelagic) or larger taxonomic groups (invertebrates, macroalgae, sharks and rays, tunas, Miscellaneous groups, Table 1). Because of their large variations in landings, mackerel and horse mackerel could mask trends of other species so some of the analyses have been conducted with and without these two species.

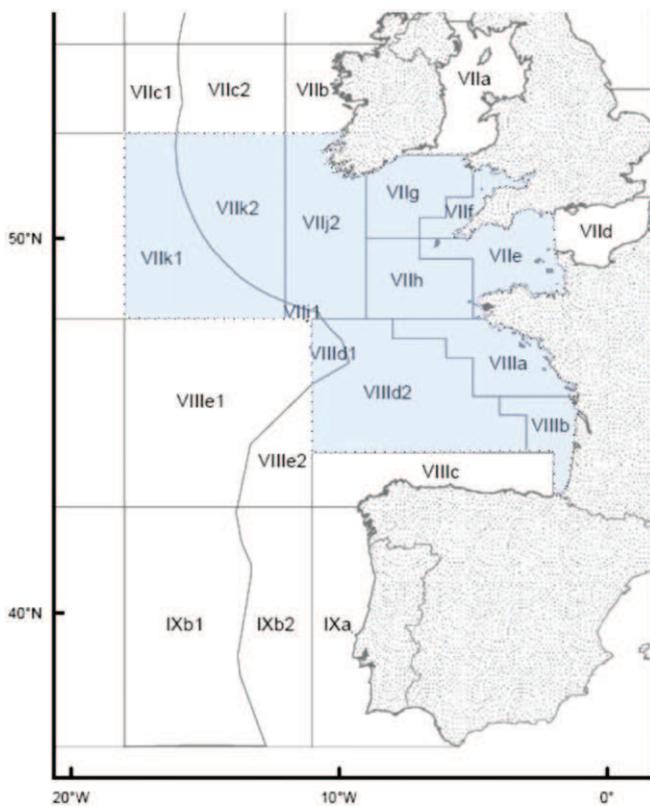
### 2.2. Stock status and trends

The effect of fisheries on the assessed populations was evaluated by examining trends in abundance, indices of stock status, and level of exploitation as characterised by the ratio F/M (fishing and natural instantaneous rates of mortality per year, Tables 2 and 3) and the comparison of the structure of landings in age and length with the age and length at maturity, and the minimum legal length (MLS).

**Table 1**

Trophic level and asymptotic length ( $L_{\infty}$ ) for fish and invertebrates that constitute the main species and miscellaneous groups.  $L_{\infty}$  is the asymptotic length of the von Bertalanffy growth equation used for fish only. Macroalgae were not included in the calculation of mean trophic level of the landings.

English name	Latin name	Trophic level	$L_{\infty}$ (cm)
<b>Main species</b>			
<i>Fish</i>			
European anchovy	<i>Engraulis encrasicolus</i>	3.11	25
European pilchard	<i>Sardina pilchardus</i>	3.1	22
Atlantic mackerel	<i>Scomber scombrus</i>	3.65	37
Whiting	<i>Merlangius merlangus</i>	4.37	49
Atlantic horse mackerel	<i>Trachurus trachurus</i>	3.64	45
Atlantic cod	<i>Gadus morhua</i>	4.42	126
Common sole	<i>Solea vulgaris</i>	3.13	54
European plaice	<i>Pleuronectes platessa</i>	3.26	82
Haddock	<i>Melanogrammus aeglefinus</i>	3.6	68
Hake	<i>Merluccius merluccius</i>	4.5	105
Herring	<i>Clupea harengus</i>	3.23	35
Sprat	<i>Sprattus sprattus</i>	3	13
Monkfish	<i>Lophius piscatorius</i> , <i>L. budegassa</i>	4.45	131
Megrim	<i>Lepidorhombus whiffiagonis</i>	4.24	59
<i>Lobsters/crabs</i>			
Bay lobster	<i>Nephrops norvegicus</i>	2.6	na
Common lobster	<i>Homarus gammarus</i>	2.6	na
Spiny lobster	<i>Palinurus elephas</i> , <i>P. mauritanicus</i> (secondary sp)	2.6	na
Edible crab	<i>Cancer pagurus</i>	2.6	na
Spider crab	<i>Maja brachydactyla</i>	2.6	na
<b>Miscellaneous groups</b>			
Sharks L	e.g. <i>Prionace glauca</i> , <i>Lamna nasus</i>	4.3	407
Sharks and rays	e.g. <i>Dipturus batis</i> , <i>Squalus acanthias</i>	3.9	213
Albacore	<i>Thunnus alalunga</i>	4.3	135
Other tunas	mainly <i>Thunnus thynnus</i> and <i>Xiphias gladius</i>	4.2	294
Pelagics M	e.g. <i>Scomberesox saurus</i> , <i>Spondyllosoma cantharus</i>	3.4	42
Pelagic L	e.g. <i>Sarda sarda</i> , <i>Sebastes marinus</i>	4.3	82
Demersal L	e.g. <i>Molva molva</i> , <i>Zeus faber</i>	3.9	116
Demersal M	e.g. <i>Trisopterus minutus</i> , <i>Micromesistius poutassou</i> , <i>Mullus surmelutus</i>	3.4	42
Demersal S	e.g. <i>Mullus barbatus</i> , <i>Apogon imberbis</i>	3.4	18
Cephalopods	e.g. squids, octopus, cuttlefish	3.4	na
Other invertebrates	e.g. scallops, urchins, clams	2.3	na
Other fish	sturgeon, salmon, unidentified	3.5	na
Macroalgae	mainly Laminaria		



**Fig. 1.** Map of the study area (delimited with a heavy line), the Celtic Sea and the Bay of Biscay.

**Table 2**  
Population dynamics characteristics, minimum legal size (MLS), and the structure of the landings in terms of percentage of immature, and stock status (using ratios F on precautionary points Fpa, Fmax F0.1) for some demersal fish and crustaceans. Lmat, Age mat: Length and age at maturity; Mesh: indicates changes in mesh size, used; F and M: instantaneous fishing and natural mortality rates respectively; years considered are for the age structure and maturity of the catch only. The estimates of F come from different terminal years (see comment *m*).

Group <i>a</i>	Max age <i>l</i>	Lmat. (year)	Age mat. (year)	MLS (cm)	Mesh <i>g</i> (mm)	M (/year)	F <i>m</i> (/year)	Years considered	% immature		% older <i>r</i>		Sources	F/F0.1	F/Fmax	F/Fpa
									Length	Age	Catch	YPR				
Cod <i>k</i>	25	53	2–3	35		0.2	0.6	2006	39	50	1.4	16	1,3,4	2.5	1.6	0.88
Sole																
Villab	26	26.5	3	24		0.1	0.33	2006	6	21	17		1,3,4,19	2.8	0.8	1.25
Celtic VIIIg		26 <i>i</i>	3	24		0.1	0.19	2006	1	10	15	33	3,4,10	1.5	0.7	0.52
Vlle						0.1	0.25						3			
Plaice VIIIg	30	28.8	3	27		0.12	0.41	2006	30	27	13		1,3,8	2.5	1.2	0.97
Haddock	20	30	2	30		0.2	0.71	2006	5	10	0.8	5.7	1,3,4	1.1	0.6	–
Hake	12	47	4.2	27		0.2	0.23	2006	66	52	13	38	4	2	1.2	0.85
Monkfish																
<i>L. piscatorius</i>	24	60	6	34 <i>c</i>		0.15	0.24	2006	82	62	5		1,4,9,19,11	4.8	3	1.08
<i>L. budegassa</i>		65	7	34		0.15		2006	94	75	8		4,9,10,21			
Megrim	15	28	4	25		0.2	0.38	1990–1999	46	–	26		3,4,10	4.4	2.8	1.46
				20				2000–2006	57	–	1		4			
Whiting						0.2	0.48						4,20			
Norway lobster VIII <i>b,e</i>		2.5	2–3	2	50	0.3	0.38	1987–1989	26	–	0.65		3,4			
					55			1990–1999	10	–	0.03		3,4			
					70			2000–2004	7	–	0.02		3,4			
				3				2005–2006	3	–	0.4		3,4			
European lobster <i>b</i>		9.7		8				before 1981					12			
				8.5				before 2002	98 <i>q</i>				6			
				8.7				2002					6			
Edible crab <i>b</i>	25	6.7–7.3 <i>j</i>		8				before 1981					5,13			
				9									5			
				13–14 <i>k</i>									14			
Spider crab <i>d</i>	7–8	~14	2–3 ?	12		0.3			10 <i>q</i>				6			
Spiny lobster <i>b</i>	~20	9.5		8				since 1961					7,15			
				9.5				since 2001					15			

Notes: *a*: study area unless otherwise indicated; *b*: Length of cephalothorax (Lc) used; *c*: Minimum weight set at 500 g which amounts to about 34 cm using  $W(g) = 0.218 * L(cm)^{2.83155}$  (Guillou, 1978); *d*: carapace length; *e*: 20 mm Lc = 7 cm total length and Lt of 9 cm = Lc of 30 mm; *f*: Glénans region in northern Bay of Biscay was deemed highly exploited, less so than Belle-Ile in the southern Bay, sampling done with a 25 mm mesh trawl; *g*: followed for Nephrops only; *h*: for females; *i*: assumed to be equal to that of Bay of Biscay due to large variations among sources and given the same age at maturity; *j*: 13 and 14 cm south and north of the 48th parallel respectively; *k*: percent specimens younger than age at maturity calculated assuming that 60% of 2 years old were immature; *l*: from FishBase (Froese and Pauly, 2012); *m*: for the most recent assessment, 2004 for monkfish and megrim, 2006 for cod, 2007 for hake, and 2009 for the others; *q*: see text; *r*: older fish defined as fish older than age at maturity +2. Sources: 1. Fishbase; 2. assumed equal to that of west Scotland (Jennings et al., 1998); 3. ICES, 2008; 4. ICES, 2007c; 5. Latrouite, 1989; 6. Forest, 2001; 7. Latrouite and Noël, 1997; 8. Jennings et al., 1998; 9. ICES, 2005; 10. Mellon et al., 1994; 11. Guillou, 1978; 12. Latrouite et al., 1981; 13. Autissier et al., 1981; 14. Latrouite 2002 ifremer 2002 fiche 32; 15. Latrouite and Lazure, 2005; 16. Letaconnoux, 1951; 17. Guillou, 1978, table 76; 18. Fontaine, 1967; 19. Ifremer, 2001; 20. ICES, 2010d.

We summarised the current status of the assessed stocks, using three reference points for biomass and fishing mortality: the precautionary reference points for biomass and fishing mortality (Bpa and Fpa), the point at maximum yield-per-recruit (Bmax and Fmax), and at F0.1 (B0.1 and F0.1) (Cadima, 2003; Hauge et al., 2007). Bpa and Fpa are the precautionary reference points commonly used by ICES working groups to avoid recruitment overfishing (i.e. Bpa and Fpa are set up from retrospective analysis and theoretically define the limits beyond which recruitment will be affected by the fishing-induced decrease in the spawning biomass). The two other reference points, F0.1 and Fmax, are calculated from a yield-per-recruit model. Fmax is the point of maximum yield and F0.1, more conservative, is the fishing mortality at which the slope of the yield-per-recruit is a tenth of the slope of

the curve at its origin, and is commonly used to define the limit of full exploitation.

All these reference points were found in stock assessment reports or could be computed from the yield-per-recruit table found therein (Table 2). For monkfish and megrim the assessment has been rejected by the advisory committee of ICES and although the trends are likely valid, the absolute values for reference points may not. We assume here that the ratio between observed F and the reference points is valid, based on the trend in biomass. In the case of hake, the assessment has recently been changed considerably and suggests higher productivity than in the previous age-structured assessment. Unfortunately, this assessment has not been carried out for the period before 1990 and is thought to be marred with lack of contrast and large uncertainties, notably for the trends in biomass. We thus continued to use the previous assessment although this may result in pessimistic estimates for the very last years.

We presented these results by adapting the synoptic method developed by Garcia and De Leiva Moreno (2005) in which the current F is compared to reference points (here Fmax and Fpa) by normalising with these two reference points. The normalised value is not proportionally related to the original observed F and B, but translates the distance to the precautionary points. Thus, the relative F (Frel) is obtained with:  $Frel = (F_{current} - F_{max}) / (F_{pa} - F_{max})$  while the relative biomass is  $Brel = (B_{current} - B_{pa}) /$

**Table 3**  
Natural mortality and fishing mortality for the last year of assessment considered for species not listed in Table 2.

Species	M (/year)	F (/year)	Year	Source
Anchovy	1.20	0.07 <sup>a</sup>	2006	(ICES, 2010c)
Herring	0.33	0.09	2008	(ICES, 2010a)
Mackerel	0.15	0.30	2005	(ICES, 2007a)
Horse mackerel	0.15	0.24	2008	(ICES, 2010e)

<sup>a</sup> fishery was minimal in 2007 and closed in 2008 and 2009.

( $B_{max} - B_{pa}$ ). Trends in stocks groupings were obtained by using a geometric mean (which decreases the weight of exceptionally large values) for all years for which data were available.

### 2.3. Ecosystem indices

In the study area, French survey data are only available for the last few years of the study period. Thus, long-term global changes were investigated using several indices estimated from landings. The mean trophic level ( $TL_y$ ) of the catch for year  $y$  (Pauly et al., 1998), defined as:

$$TL_y = \frac{\sum_i (TL_i \cdot Y_{iy})}{\sum_i Y_{iy}}$$

where  $Y_{iy}$  is the landings (yield) of year  $y$  for species or group  $i$ , and  $TL_i$  is the trophic level of species or group  $i$ . The mean trophic level of total landings excludes macroalgae. Catch trophic spectra (Gascuel et al., 2005) were estimated, on average per decade, from landings and TL by species or group. For each species or group, landings are distributed on a range of trophic levels in order to take into account the variability of TLs, using the smoothing procedure detailed in and based on small trophic classes of width conventionally fixed equal to 0.1 TL. The catch trophic spectrum is the distribution over trophic levels of the total catch, obtained by summing distributions for all species or groups.

The fishing in balance index (FiB, Pauly et al., 2000) was used to determine whether decreases in trophic level were matched by increases in landings,

$$FiB_y = \log(Y_y \cdot TE^{-TL_y}) - \log(Y_0 \cdot TE^{-TL_0})$$

where  $Y_y$  is the landings (yield) of year  $y$ , TE the trophic efficiency set at 0.1 (Pauly and Christensen, 1995),  $TL_y$  and  $TL_0$  are the mean trophic level of the catch of year  $y$  and of the first year of the time series respectively. The mean asymptotic length  $L_\infty$  of year  $y$  of landings was estimated as an additional index of changes in landing composition and computed as:

$$L_\infty_y = \frac{\sum_i (L_{\infty_i} \cdot Y_{iy})}{\sum_i Y_{iy}}$$

The primary production required for landings (PPR) (Pauly and Christensen, 1995):

$$PPR_y = \sum_i Y_{yi} \cdot 10^{TL_y - 1}$$

Trophic level and asymptotic length ( $L_\infty$  in the von Bertalanffy equation) were taken from Fishbase (Froese and Pauly, 2012) for all fish species (Table 1). Trophic level values for miscellaneous fish groups were calculated based on the species composition. Trophic levels for invertebrates were taken from a Bay of Biscay Ecopath model (Ainsworth et al., 2001).

### 2.4. Trends in the French CPUE

We examined the trends in landings and effort for the fleet based in French ports of the Atlantic coast and western Channel for the period 1950–2006 (year 2007 and 2008 were not available). Since the fishing effort does not refer to a specific area, we first considered the French landings coming from the study area and then used all the French landings coming from the West European waters, which is more representative of the  $E_{corr_y} = KW_y \cdot (1 - \alpha)^{n-1}$  activity of the fleet. The West European waters are defined as all Atlantic waters of

the ICES region (divisions V–XII) excluding the eastern Channel (VIId) (Fig. 1).

Effort for the French Atlantic region fleet (Bay of Biscay and western English Channel ports), expressed as the sum of engine power in kilowatts (KW), was compiled for years 1950–2006 by combining various sources of information. Starting with the time series published for all aggregated French regions published by Mesnil (2008), we derived effort of French fleets for the Atlantic region. Between 1950 and 1982, fishing effort located on the Atlantic coast constitutes 64% of the total French effort, based on the observed ratio of Atlantic ports (Boloipon et al., 2000) and SIH (Système d'Information Halieutique) data for France (Leblond, Ifremer Nantes, pers. comm.). The effort for 1999–2006 was estimated at 75% of the Northern France (Atlantic plus eastern Channel), based on the ratio observed in 1998. It is well known that fishing power is not only linked to the engine power but also results from the use of hydraulic systems, high-performance fishing gears, number and size of fishing gears, better locating systems (Biseau et al., 1984), or 3D mapping sonars (ICES, 2007b), all of which contribute to increase fishing efficiency. Assuming a mean increase in fishing power of  $\alpha\%$  per year the corrected fishing effort ( $E_{corr}$ ) in any giving year  $y$  is:

$$E_{corr_y} = KW_y \cdot (1 - \alpha)^{n-1}$$

where  $n$  is the number of years before 2006 and  $\alpha$  the mean yearly rate of increase in fishing power (Gascuel et al., 1993). We first considered a mean value of  $\alpha = 2\%$  per year and also considered extreme values ( $\alpha = 1$  and 3%) which are assumed to be sufficient to include the true mean of the average yearly increase in the fishing power of each KW in the European context. These values will be discussed further in Section 4. Note that an  $\alpha$  of 2% means that the efficiency of each horse power doubles after about 35 years, while a value of 3% implies doubling efficiency every 24 years. The catch per unit effort (CPUE) based on this corrected effort was rescaled so that the CPUE would be equal to 100 in the last available year.

## 3. Results

### 3.1. Landings

Total landings in the study area have slightly increased from 438 to 508 thousand tonnes between 1950 and 2008 (Fig. 2a). French landings increased also but more regularly and constitute an average of 40% of international landings in the study area. The large international landings observed in the mid-1970s and 1990s are due to mackerel (*Scomber scombrus*) and horse mackerel (*Trachurus trachurus*), which dwarfed all other pelagic species before decreasing during the last decade (Fig. 2b). Globally, and excluding these two species, total landings remains rather stable since the late 1950s. Some heavily fished species like hake (*Merluccius merluccius*), anchovy (*Engraulis encrasicolus*) or tuna (mainly *Thunus albacares*) decreased markedly (Fig. 2c and d). This decrease was compensated by the exploitation of a larger part of the ecosystem. Exploitation of new species like monkfish (*Lophius piscatorius* and *Lophius budegassa*) started as the market developed in the 1970s and biomass increased. Similarly, cephalopods and large crustaceans landings, mostly edible crab (*Cancer pagurus*), and spider crab (*Maja brachydactyla*) increased markedly starting in the 1970s. The composition of Miscellaneous landings has changed markedly over time (Fig. 2d), due in part to improvements in reporting of the "other fish" category and to the increase of cephalopods and various invertebrates.

For some emblematic species longer times series can be rebuilt (Fig. 3). Between the wars, hake landings oscillated between 20 and

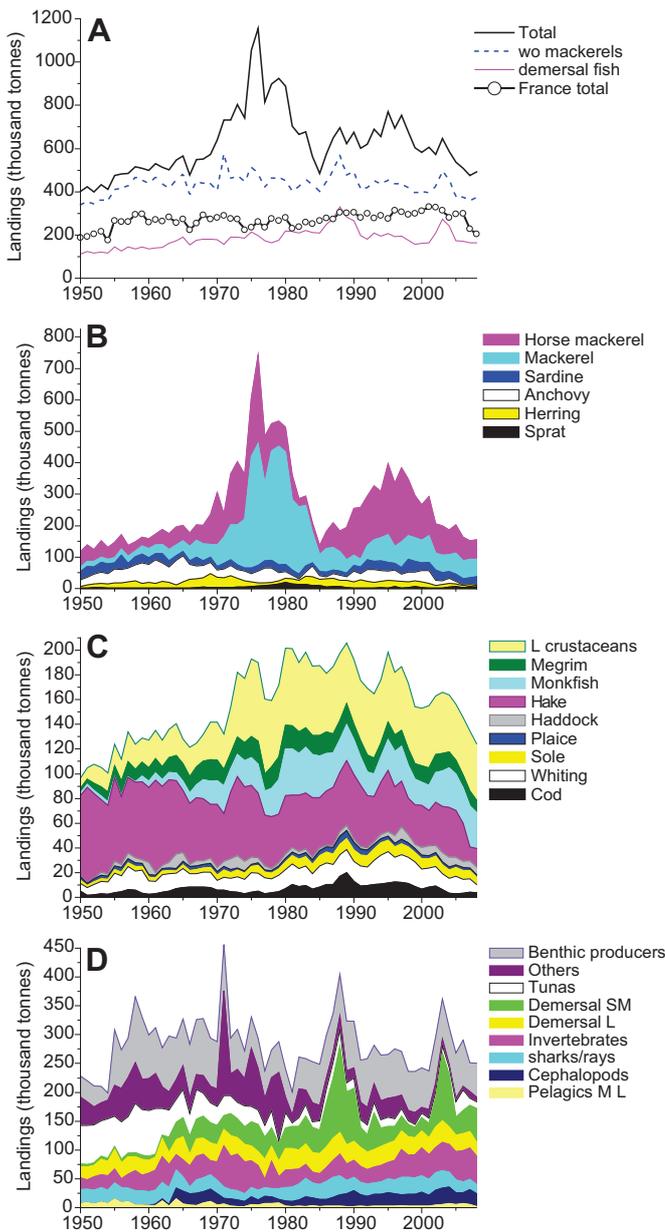


Fig. 2. French landings and international landings for the study area showed for all groups (A), pelagics (B), demersals (C) and miscellaneous (D) groups. 'wo mackerels' = landings excluding mackerel and horse mackerel.

40 thousand tonnes. In 1947, however, landings reached 120 thousand tonnes, about 4-fold the pre-war landings and decreased rapidly subsequently from 65 thousand tonnes in 1950 to 31 thousand tonnes in 2006. In 1946, trawlers' CPUE observed in La Rochelle, the most important port for this species, was two to four times higher than in 1937–1939 (Letaconnou, 1948), and already back to pre-war levels in 1949 (Letaconnou, 1951).

Cod landings reached over 6000 t in 1921, after the First World War, and declined rapidly to an average of about 3000 t (Fig. 3). After the Second World War, landings varied from 6700 t to 11,000 t until 1989 where it reached an exceptional value of 20,000 t due to a few abundant cohorts. Cod CPUE, estimated at a yearly average of 3 kg per trip (max. of 18 kg in August) before the second World War (1937–1939), increased to 334 kg per trip (max. 2099 kg in July) in 1946 (Letaconnou, 1948) which could be attributed to the absence

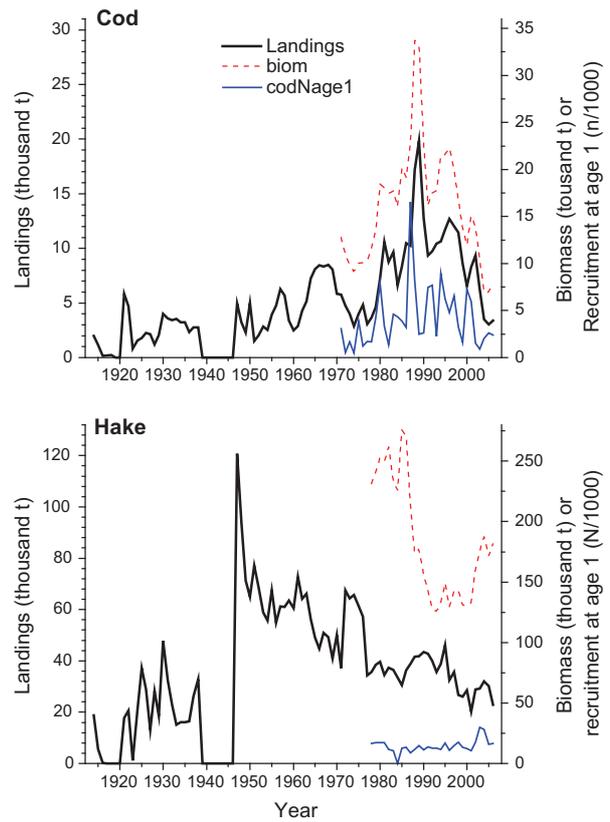


Fig. 3. Hake and cod landings since 1914, and biomass and recruitment estimated in stock assessments.

of fishing coupled with the possible occurrence of large cohorts during the war. This suggests that the fishery was already important before the war compared with the population size. Unfortunately, no information on size or age structure was available for that period. During the last two decades landings and biomass markedly decreased and the stock is currently considered one of the most overexploited in the Celtic Sea. In 2008, yearly landings (3639 t) were lower than those observed before or just after World War II.

### 3.2. Succession of species, the case of crustaceans

The exploitation of large crustaceans is a good illustration of the protracted succession of species targeted (Fig. 4) and fishing grounds exploited, and also of the remarkable increase in fishing pressure. The sudden increase in landings of large crustaceans in the 1970s is due to the exploitation of two new species: spider crab (*M. brachydactyla*) and edible crab (*C. pagurus*) (Fig. 4).

In France, the interest in these species resulted from the decline of European lobster (*Homarus gammarus*), spiny lobster (*Palinurus elephas*), and the Bay of St. Brieuc common scallops (*Pecten maximus*) between 1958 and 1970 (Autissier et al., 1981; Latrouite and Lazure, 2005). Spider crab landings started in 1973, peaked in 1977 at 12,000 t and stabilised at an average of 5600 t in the last 10 years (Fig. 4). The development of the fishery was accompanied by an impressive increase in fishing capacity. For instance, in Saint-Malo, the most important fishing port for spider crab, the length of nets used increased from 3 km per boat in 1973 to 20–25 km in 1977 and the mesh size decreased from 160 mm to 110 mm (Autissier et al., 1981). The fishing season for this species was extended, and netters started exploiting offshore fishing grounds.

The existing fisheries have gone through their own cycle on development. For instance, the fishery for spiny lobster started as

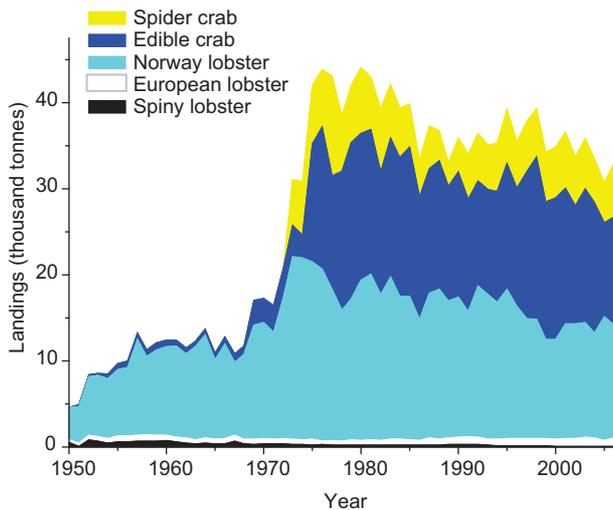


Fig. 4. All countries landings for large crustaceans in the study area.

early as the 19th Century and developed when the sardine fishery declined (1902–1913) (Latrouite and Lazure, 2005, appendix 5). Already then, decreases in yield on the nearest fishing grounds pushed fishermen to seek further fishing grounds from Scotland to Africa while the number of boats increased considerably. Landings in the study area declined from a yearly average of 573 in the 1950s to an average of 75 t in the 2000s (Fig. 4). Similarly, Norway lobster (*Nephrops norvegicus*) fishing already existed in the 1940s in the Bay of Biscay. Due to the development of the trawl fishery, and possibly to an increase in the population size, landings of Norway lobster increased from less than 5000 t in 1950 to more than 12,000 t in 1960 (Fig. 4). In the 1960s, in response to the decrease in landings and the augmented market for this product, fishermen increased their fishing effort in the fall and winter, increased the fishing capacity of their boat and started exploiting new fishing grounds (e.g. Irish Sea, English Channel, Fontaine and Warluzel, 1969). International landings for this species reached more than 20,000 t in the mid-1970s and slowly decreased to around 15,000 t nowadays.

### 3.3. Age and length structure

The length and age structure for the 14 species and stocks whose data were available show that except for flatfish, and edible crab, all fish and invertebrate species are recruited before they reach maturity (Table 2). For instance, 82% of *L. piscatorius* and 94% of *L. budegassa* landings are smaller than length at maturity reached at 6 and 7 years old respectively. Although females of the Celtic Sea cod reach maturity very early, 2–3 years old and 53 cm in length, most females will not survive their first year at maturity as fishing mortality at ages 2 and 3 reached about 0.8 in 2007. The cod population, a species that can live 20 years and reach an asymptotic length of 126 cm (FishBase, Froese and Pauly, 2012), is presently dominated by fish younger than 3 years old, while specimens older than 5 constitute only 1.4% of landings and has remained consistently below 2% since 1981. By comparison, a yield-per-recruit model predicts that older individuals (6+) would compose 16% of the landings at  $F_{max}$  ( $=0.37$ ). Similar results were obtained for the Celtic plaice and for sole, haddock, and hake.

The habit of fishing small fish is not new and hake constitutes a good example. Between 1965 and 1975, hake landed by France and Spain were as small as 5–10 cm (Guillou, 1978) whereas in 2006, landings were dominated by individuals of 30–40 cm. In all

cases, fish was largely caught before females reached the length at maturity (47 cm). The decrease in proportion of small fish in recent years is probably due to a series of management measures including the 2001 emergency management measure that increased the otter-trawlers minimum mesh size to 100 mm (ICES, 2007c). The procedure to obtain the respect of the minimum size limit (27 cm) was long and culminated in 2006 with France being condemned to pay a large fine (57.7 million Euros) by the European Union (ICES, 2007c).

### 3.4. Fishing mortalities and stock status

Of the 16 stocks considered, 8 (flatfishes, cod, haddock and whiting) were consistently fished at more than two times the natural mortality ( $M$ ) of which 5 reached  $F/M$  ratio higher than 6 during the period of the stock assessment (Fig. 5, Tables 2 and 3).

The ratio  $F/M$  remained below 3 for hake, megrim, mackerel, herring, Norway lobster, and monkfish, and below or close to 1 for anchovy and horse mackerel. The ratio is quite low for anchovy because of the high natural mortality of 1.2 compared to the low  $F$  estimated during (0.67 in 1987 to 0.07 in 2006). The current  $F$  (signalled by diamonds in Fig. 5) is close to the minimum observed during assessment years for 8 stocks.

The results may prove to be pessimistic for a species like hake in the future in light of the possible higher productivity. Nevertheless, the mean ratio  $F/F_{max}$  for cod, Celtic plaice and sole, and hake, increased from 1.1 to 2.2 between 1978 and 1990 and declined starting in 2003 (Figs. 6a and 4 stocks).

The mean biomass ratio  $B/B_{max}$  decreased from 1.2 to 0.54 between 1985 and 2006. Adding Bay of Biscay sole and monkfish produced similar trends (Fig. 6a, 6 stocks). The recent decrease in fishing mortality has not resulted in a corresponding increase in biomass yet.

For all assessed species, the level of fishing mortality estimated in 2008 (or the last assessment year available) is close or higher than  $F_{pa}$  (except for the Celtic sole), higher than  $F_{max}$  (except for soles and haddock) and higher than  $F_{0.1}$  (Table 2). Reaching  $F_{0.1}$  would necessitate a 2-fold reduction of fishing mortality for hake, cod, Bay of Biscay sole and the Celtic Sea plaice, and more than four-fold for monkfish, megrim. On average, stocks that are already fished at an  $F$  higher than  $F_{max}$  would require a 2-fold decrease in fishing mortality (Table 2). Since the mid-1980s, the relative  $F$  ( $F_{rel}$ ) increased beyond  $F_{pa}$  for cod, hake and the Biscay and Celtic soles while  $B_{rel}$  declined below  $B_{pa}$  and hence, the stocks were in the 'dangerous zone' for several years (Fig. 6b dashed line). Only in 2006 has  $F$  reached a level below  $F_{pa}$ . Adding megrim and

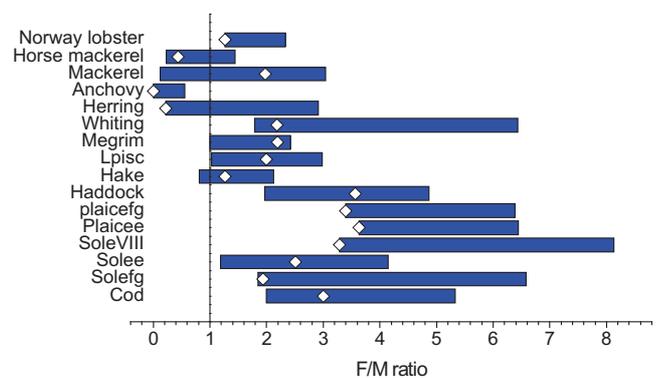
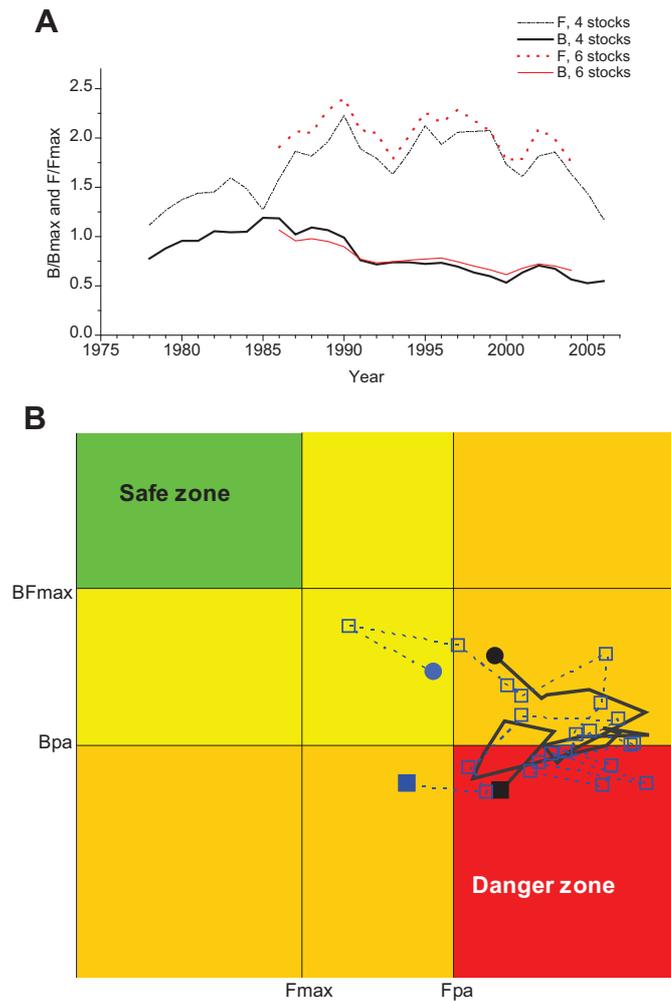


Fig. 5. Range of yearly  $F/M$  ratio observed during the period covered by stock assessments. The diamonds represent the last year of the stock assessment. Monkfish is represented by only one species here, *Lophius piscatorius*.



**Fig. 6.** Status of available stocks, comparing current biomass and fishing mortality with the  $F_{max}$  reference point (A); and relative biomass and fishing mortality compared to the precautionary ( $F_{pa}$ ) and maximum yield-per-recruit ( $F_{max}$ ) thresholds. A. 4 stocks are cod, Celtic sole and plaice, and hake; 6 stocks: plus Biscay sole and monkfish. B. dashed line and open squares: hake, cod, Celtic sole and plaice for 1984–2006; solid line: plus monkfish and megrim from 1986 to 2004. The beginning of a time series is signalled with a filled circle and the end with a filled square.

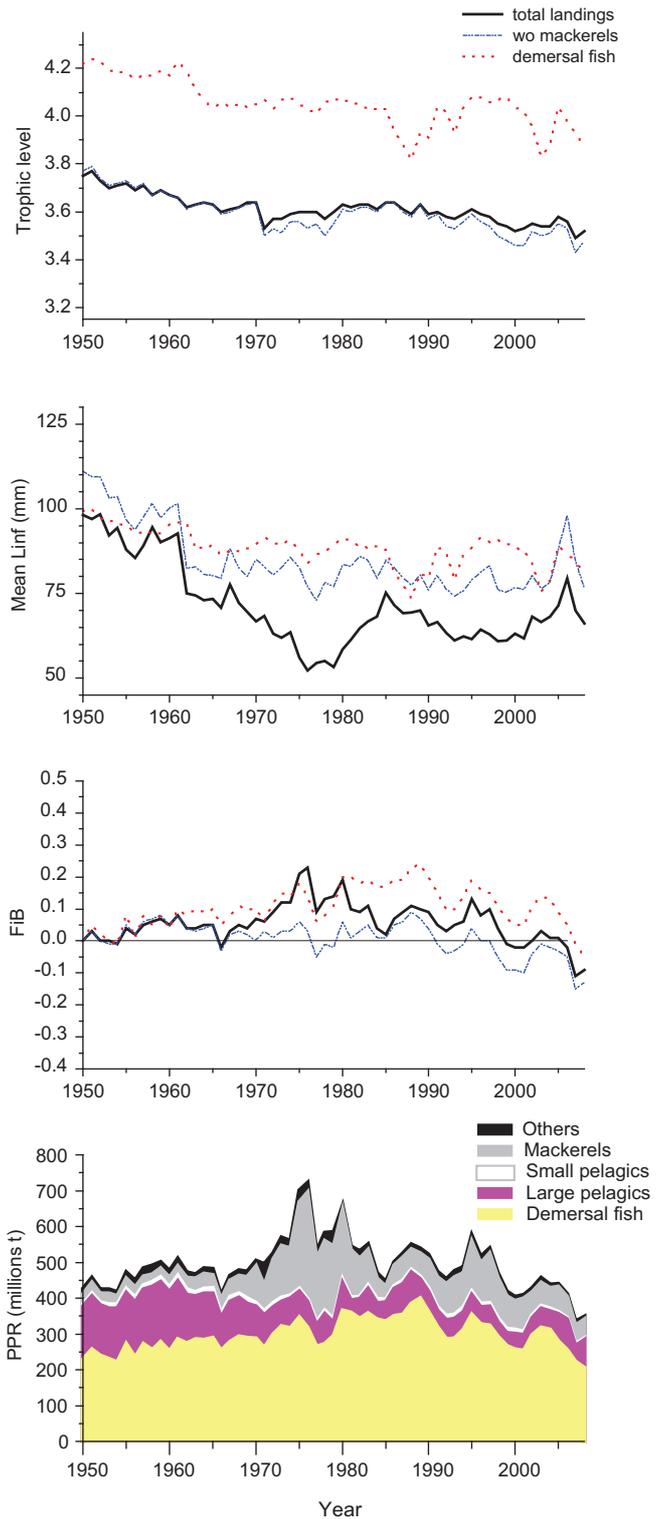
monkfish to the groups resulted in basically the same trend although  $F_{rel}$  was consistently larger than  $F_{pa}$  (Fig. 6b solid line).

3.5. Ecosystem indices

Ecosystem indices show a change in landing structure and composition over the study period. The mean trophic level of total landings declined from 3.75 to 3.52, a rate of 0.03 TL per decade, and at a steeper rate of 0.08 TL/decade between 1950 and 1970 (Fig. 7a).

The rate of decline decreased in the 1980s, due to the augmentation of landings in monkfish and cod which partially counterbalance the increasing importance of lower trophic levels groups, small and medium sized demersal and invertebrates. When mackerel and horse mackerel were excluded from the calculations, the trophic level of landings decreased from 3.77 to 3.48 (Fig. 7a). The trophic level of demersal fish considered separately was at 4.2 in the 1950s and declined at a rate of 0.044 TL per decade.

The decrease in TL was accompanied by a decrease in the mean asymptotic body length. Total landings, excluding invertebrates and



**Fig. 7.** Trends in mean trophic level, mean fish asymptotic length ( $L_{\infty}$ ),  $F_iB$  and PPR between 1950 and 2006 for landings of all species (except macroalgae), excluding mackerel and horse mackerel (wo mackerels), and demersal fish.

the group “other fish”, sharply declined in mean length in the 1960–1970s as tuna landings decreased (Fig. 7b), and was brought down further by landings of mackerels in the 1970s and 1990s. The mean asymptotic length decreased from an average of 98 cm in the 1950s to 66 cm in the 2000s. The mean asymptotic length of

demersal fish decreased from an average of 99 cm in the 1950s to 81 cm in the 2000s. This decline reflects a significant change in the species composition of catches, with the presence of more small species such as whiting, sole, megrim, and blue whiting.

The fishing in balance index (FiB) generally increased during the first decades and reached its maximum value in 1976 (FiB = 0.23) (Fig. 7c). The decrease in mean trophic level of total landings, excluding macroalgae, was compensated by sudden increases in mackerel and horse mackerel landings and thus, the increase in FiB reflects the exploitation of new species, previously less or not targeted (and possibly less abundant). The FiB for demersal fish increased irregularly until 1989, the decline in mean trophic level being more than compensated by the increase in landings (including newly targeted species such as monkfish). The index declined over the last 3 decades (from 0.20 in the 1980s to 0.06 in the 2000s) due to the decrease of both catch and mean trophic level. For all FiB indices, the trend declined over the last decades, highlighting that the development phase has passed. This suggests that the release of predation (potentially resulting from the overfishing of high-TLs) is not sufficient anymore to induce an increase in the abundance and catch of prey, which are also heavily fished. It could thus benefit, at least partly, to unfishable species.

The PPR indices show the increasing use of primary production required by fisheries until the 1970s for the whole ecosystem and until the late 1980s for demersals (Fig. 7d). Such a trend results from the increasing fishing pressure at the ecosystem level and a larger portion of the resources being progressively targeted. The PPR for demersal resources decreased after 1990, following the decrease in mean TL and landings and this can be interpreted as an overexploitation of the ecosystem's demersal compartment. Also, the exploitation of large pelagics, requiring about two-thirds of the total PPR in the 1950s, declined sharply in the 1960s and remained low afterwards. Large pelagics includes high trophic level species such as sharks and tunas (mainly albacore and bluefin), whose landings have been declining for a long time.

The decadal trophic spectra computed for all species and groups landed (including invertebrates) highlight again the main changes that have occurred during the study period (Fig. 8).

In the 1950s, the fishery exploited mainly species of high trophic levels (>4), while small landings were reported for trophic level groups of about 3. During the following decades, landings of the highest trophic levels (>4) continuously decreased while those of the lowest levels increased, following the increasing landings in

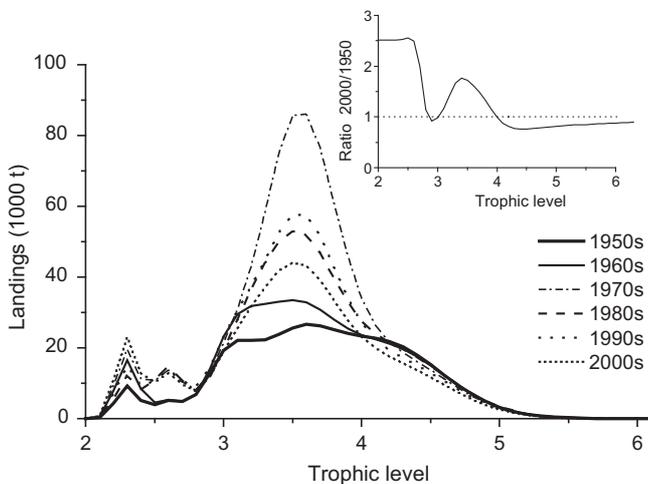


Fig. 8. Decadal trophic spectra of the landings in the study area and ratio 2006/1950 by trophic level in inset.

large crustaceans and small benthic species over time (Fig. 2). The landings of mean trophic level of about 3.5 have increased in the 1970s and decreased in the 1990s following the mackerel's relative importance. By 2000, landings of trophic level groups 2–2.5 are 2.5 times greater than in the 1950s, TL of 3.5 were 1.7 times more important, while trophic levels larger than 4.0 represent only 83% of what they were in 1950 in spite of a huge increase in fishing effort.

### 3.6. French effort and CPUE indices

Nominal effort increased regularly between 1950 and 1990 from 164 thousand to 748 thousand KW and then decreased to 503 thousand KW in 2006 caused by the decommissioning of parts of the fleets (Fig. 9).

The relative value of 1950 effort expressed in terms of 2006 kW is uncertain and depends on the assumed rate of increase in efficiency. A yearly rate of increase of 2% means that the realised fishing pressure in 1950 would have been of only 10.4% of the fishing pressure present in 2006. In other words, based on our medium hypothesis, the fishing pressure would have been multiplied by 10 over the study period. CPUEs based on effective efforts can be considered an index of the whole abundance, for all exploited species. Depending on the scenario of increase in fishing power, these indices declined 3-fold–10-fold during the study period, with a 6-fold decrease for the mean scenario. The key point is that all indices decreased rapidly from the 1950s to 1970s while

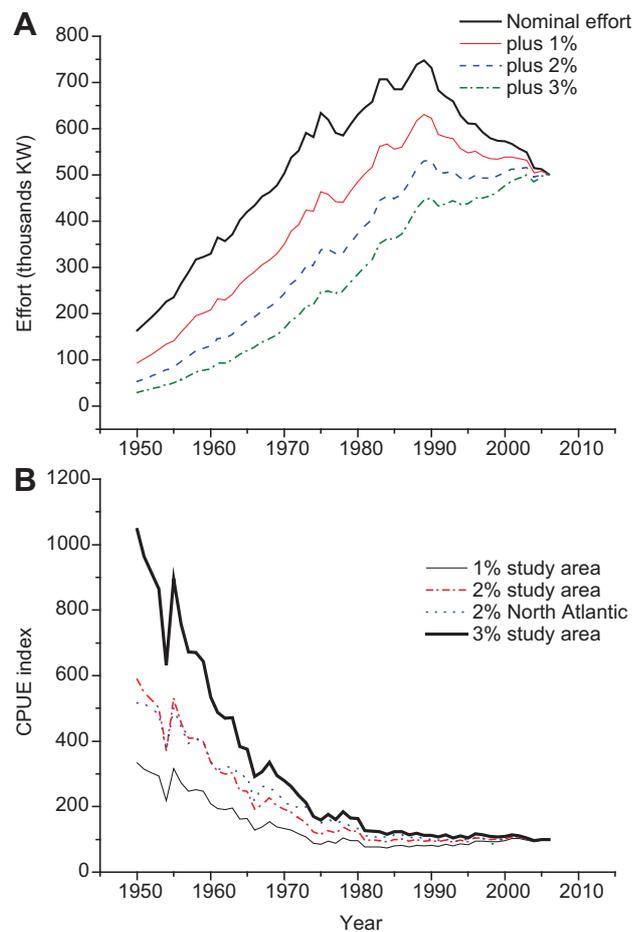


Fig. 9. Nominal effort in KW and scenarios of rate of increase in fishing efficiency (1, 2 and 3%) (A), and the corresponding CPUE (B).

abundance indices appeared very stable after 1980 for all scenarios. The decline in abundance is similar whether we consider the landings from the study area or the western European waters. Thus, all scenarios yield low mean abundances for more the last 30 years.

## 4. Discussion

### 4.1. Stocks overexploitation

Judging from the level of landings and the rapid decline in CPUE documented for several species, the Celtic Sea and Bay of Biscay ecosystem has been intensively fished starting in the 1950s at least and the exploitation was not negligible before World War II. For instance, hake landings were similar to what they are today while cod landings were similar before and after the war. Both species benefited from the respite and showed an increase in CPUE after the war as did most other species (Letaconnoux, 1948).

Most exploited species considered are characterized by a severe truncation of their length and age structures, the reliance of the fishery on new recruits, and a large proportion of immature individuals in the landings. Also, fishing mortality was generally appreciably larger than  $M$  except for pelagic species (whose  $M$  is often higher) and some demersal species (in the most recent year of assessment). Similar status of exploited species have been found in the Baltic and North Seas (Froese et al., 2008), and the Mediterranean Sea (Stergiou et al., 2009).

Froese et al. (2008) showed that current exploitation rules (minimum legal size smaller than length at maturity and smaller than the optimal size,  $F = 2-6 * M$ ) resulted in serious over-exploitation and growth overfishing for 6 demersal stocks of the Baltic and North Seas. Also, a more sustainable catch could be achieved by decreasing fishing mortalities to a value ranging from  $0.6 * M$  to  $M$ . Higher catches would eventually be reached if length at first capture were increased to optimal length while at the same time the stocks biomass on average seven fold increased. Garcia et al. (2012) suggested that unselective fisheries would help maintain biodiversity and ecosystem structure providing that the average fishing exploitation for the entire ecosystem was considerably decreased. However, the practical impact on sensitive species was not explored in the paper.

For demersal fish, Walters and Martell (2004) recommended fishing mortality no higher than  $0.8 * M$  and even  $0.6 * M$  when the length at first capture is low. Patterson (1992) recommended  $F = 0.6 * M$  for pelagics. Although the multi-species MSY curve tends to have a flat peak that suggests wide ranges of exploitation rates around the 'maximum sustainable ecosystem yield' (Hilborn and Stokes, 2010), the consequences in terms of collapsed species and ecosystem structure can be important (Froese and Proelß, 2010; Froese et al., 2008; Worm et al., 2009). Finally, on an ecosystem basis, it is probably fair to expect an erosion of the ecosystem if all species including forage fish were exploited at  $F_{msy}$  (Smith et al., 2011; Walters et al., 2005). These recommendations are quite lower than observed fishing mortalities and lower than the precautionary approach reference points.

The so-called precautionary approach, based on  $B_{pa}$  and  $F_{pa}$  targets, was formally adopted by ICES in 1998 to help stopping the decline of severely depleted stocks, such as hake or monkfish, and more generally for the main demersal stocks. In spite of the original intentions, the reference points have been increasingly used as targets rather than absolute lower boundaries that should not be crossed (Piet and Rice, 2004). In most cases, the current spawning biomass and/or fishing mortality index of assessed stocks are near (above or below) the threshold ( $B_{pa}$  or  $F_{pa}$ ), giving an impression of sustainability, results similar to those obtained for the North Sea (Garcia and De Leiva Moreno, 2005), the Celtic Sea (Shephard et al.,

2011), and European waters (Froese and Proelß, 2010). Nevertheless, the same comparison with  $F_{max}$  (as a proxy for MSY) highlights the overexploited status of all stocks, all located very far from the new targeted zone now delimited under the MSY principle. The optimistic view is a result of setting the reference points based on available time series, too short to be fully informative given that most stock assessments started when the stocks were already submitted to excessive fishing mortality. For instance,  $B_{pa}$  and  $F_{pa}$  set for Celtic cod ( $F_{pa} = 0.68$  or  $\sim 2 * F_{max}$ ) are related to the biomass observed in 1976, a year of relatively low abundance before the population increased due to large recruitment. As cod was already fully or overexploited at the time, the resulting reference points would likely be overly optimistic and ignore the potential recovery level of biomass for this stock. Thus reaching the MSY target, formally the new European rule agreed to at the Johannesburg summit, will require a 2–3-fold decrease in fishing pressure for most demersal stocks of the studied area (Table 1).

As a result, the ICES precautionary approach had a perverse effect in that it maintained assessed stocks at the theoretical minimum required to ensure stock renewal. For many stocks,  $B_{pa}$  is about one-third of  $B_{msy}$  (Froese et al., 2011). This implies a huge economic inefficiency for all European fishery sectors. For each stock, the same amount of catch (and even higher for many stocks) could be obtained with a much smaller fishing effort, implying lower costs and a lower ecological impact. Thus, the so-called precautionary approach resulted in the development of higher fishing pressure to catch a rare resource. Conversely, an abundant resource would afford high catches and better profitability using moderate pressure (Gascuel et al., 2011). The new  $F_{msy}$  target, now adopted by ICES should improve the management goals and help stock rebuilding.

### 4.2. An ecosystem intensively fished

Ecosystem indicators based on catch do not necessarily describe the ecosystem reliably (Branch et al., 2010; Essington et al., 2006) but they are nevertheless useful in informing on the patterns of exploitation which could admittedly result from a change in abundance or in fishing strategy, as a result of management policy or economic dynamics. They allowed us to rebuilt time series since 1950 taking into account all the exploited resources, while ecosystem indicators based on scientific observations could be estimated only since the late 1980s when surveys started and only refer to demersal resources.

Over the last 50 years, not only the fishing pressure increased but fishermen progressively exploited a larger part of the ecosystem. This includes crustaceans and molluscs, but also smaller species of fish as revealed by the very significant decrease observed in the mean asymptotic length ( $-32$  cm for all fish and  $-19$  cm for demersal fish). The history of exploitation (e.g. large crustaceans) and indices such as the decrease in mean TL of landings (0.23) and the trophic spectra attest to this change in species exploited and the increase of smaller/low trophic species. These results are consistent with the decline in trophic level found for the Celtic Sea (Pinnegar et al., 2002), the Bay of Biscay (Steinmetz et al., 2008), and the Northeast Atlantic (Pauly et al., 1998). Changes in exploitation patterns were partly caused by an increase in abundance of fish from lower latitudes in the ecosystem and in the catch, and by economic imperatives: French fleets replacing high value ground-fish with large volumes of lower value fish as global prices decreased (Steinmetz et al., 2008). The history of fishing since the 1950s shows a pattern of adding new species of both high and low trophic levels characteristic of 'fishing through' the foodweb (Essington et al., 2006). Nevertheless, the decrease in landings of high trophic level fish since the 1980s in spite of high fishing

mortality (which decreased only in the last few years) suggests that the ecosystem has also been submitted to 'fishing down' the foodweb (Pauly et al., 1998). Finally, trends in the FiB and PPR indices show that the decrease in trophic level was not completely compensated by an increase in landings, especially since the 1980s when the ecosystem became fully exploited.

#### 4.3. Using history for a better perspective

Due to data availability, CPUEs were examined only for the French fleet which accounts for 40% of the landings in the study area. Thus, in term of fishing pressure the trends are valid only for the French fleet which has benefited from numerous types of subsidies over the years (Mesnil, 2008). Although we would expect differences in trends in the fishing effort among fleets as a result of market preference and national policies, their trends in CPUE are likely to be similar resulting from the trend in the fishable biomass. Nevertheless, our estimates strongly depend on the assumption we made regarding the yearly rate of increase in the fishing power. The range we used, from 1 to 3% of yearly increased, appears reasonable and is likely to cover the true mean value of this parameter. Higher values have been estimated in some fisheries. For instance, in the Baltic Sea, estimates of fishing power increased by 2 and 6% annually for Dutch trawlers and gillnetters respectively between 1987 and 1998 (Marchal et al., 2001). For Australian shrimp trawlers Robins et al. (1998) showed that the addition of GPS and plotter system alone resulted in a 0.5–12% annual increase in fishing power for the period 1988–1992. However, these estimates usually refer to fishing efforts expressed in standardised fishing days. Assuming that technological improvements are partly linked to the increasing engine power of each boat and taking into account that our nominal effort does not include the effect of restrictions on number of fishing days that occurred in recent years, our maximum rate of increase of 3% appears reasonable.

In fact, there is no robust method to estimate the past trend in the fishing power and a large uncertainty remains regarding the CPUE estimates for the beginning of the period. Nevertheless, looking back to the 1950s and contrasting catch and effort trends highlights the impressive decline in CPUE that occurred until the 1980s, at which point CPUEs seem to stabilize. This means that looking back only 20–30 years would lead to believe that the situation is stable and the ecosystem has not changed in a typical case of 'shifting baseline' syndrome (Pauly, 1995). This would also be the case for most indicators employed in this study. In reality, the apparent stability of the last 30 years masks the fact that landings have been sustained at the cost of a dramatic increase in fishing pressure, and a change in species composition and fishing grounds. This reduction would be more pronounced if the addition in new species and fishing grounds had been taken into account.

The last 50 years have seen a spectacular increase in fishing capacity, and interestingly enough, the money and effort dedicated to increase fishing pressure have only succeeded in slowing the decline in landings while the CPUE (and perhaps profitability) (Sumaila et al., 2008) plummeted starting as early as in the 1960s. Although individual fleets are driven by profit which defines their choice in target species and fishing grounds (Sethi et al., 2010), it is also true that they have not been submitted to the full rigour of profitability in pursuing fishing activities as important subsidies have artificially maintained fleets and encouraged increase in fishing effort and fishing power (Mesnil, 2008; Sumaila et al., 2008; Villasanté, 2010).

The current fisheries management is influenced by an irresistible temptation to stretch the limits and ignore management rules to avoid yet another impending fisheries crisis (see Mesnil, 2008). The incentives to ask for exemptions have been numerous and

justified by an imminent crisis every time. In 2008 for instance, after fishermen protests, the French government requested an increase in the Celtic cod quota, arguing an unexpected recruitment, higher than the mean recruitment used in stock projections. The quota was increased in spite of the negative advice from the STECF (Scientific, Technical and Economic Committee for Fisheries, European Commission).

From this analysis, we conclude that the Bay of Biscay and Celtic Sea ecosystem shows sign of erosion because most currently assessed stocks show severe signs of overfishing, and because the landing species composition has drastically changed reflecting a long-term change in the ecosystem itself. This may be the result of a combination of additional factors such as global warming (Perry et al., 2010; Simpson et al., 2011), hypoxic water masses (Zhang et al., 2010), and the impact of fishing impact on benthic habitat structure (Kaiser et al., 2002; Simpson and Watling, 2006). A decrease in exploitation rates will be necessary to rebuild the overexploited stocks (or at least some of them) and perhaps some of the past ecosystem properties. However, given that over-exploitation is only one of the factors of change, ecosystems may never rebuild completely to those of the past. The precautionary approach formally adopted by ICES in 1998 probably contributed to stop the decline for the most affected stocks. Indeed, fishing mortality has decreased recently (this work) and the biomass has increased slightly for some stocks (STECF, 2011). This confirms that stock rebuilding can occur when proper management rules are implemented although the rate of recovery seems low. The long-term past perspective we tried to build here show how much exploited resources have been deteriorated and suggests that rebuilding a productive ecosystem will be a long process.

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#### Appendix A. Supplementary material

Supplementary material associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ocecoaman.2012.06.010>.

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