

## IMPACT OF FISHING AND CLIMATE ON THE CELTIC SEA AND THE BAY OF BISCAY<sup>1</sup>

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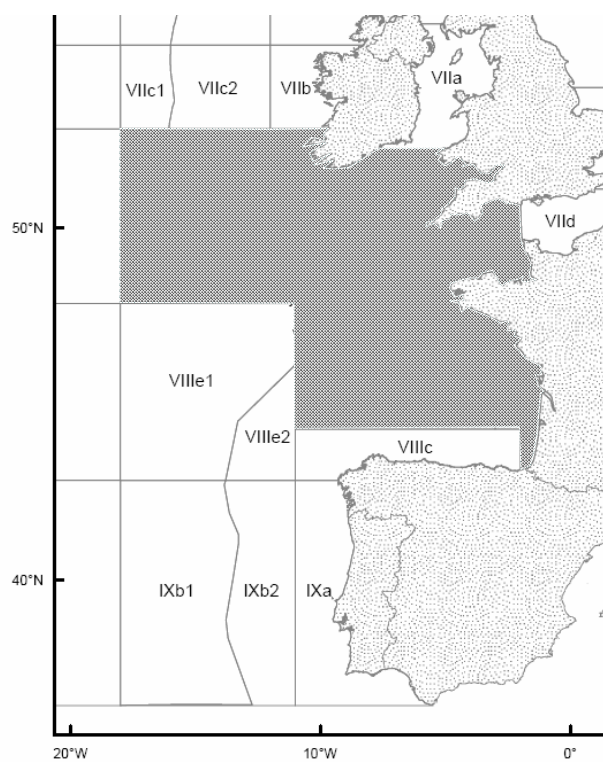
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### ABSTRACT

The Celtic Sea and the Bay of Biscay (Figure 1) have been fished intensively for at least a century. Already sizeable between the two World Wars, the fisheries resumed after 1945 with the support of governmental subventions to modernize the fleets and thus causing an unprecedented increase in fishing capacity in the region. Decrease in biomass and landings of several exploited species hint at the impact of fishing while changes in climate and productivity are also suspected. This project examines the impact of fishing on this ecosystem using an Ecosim model and taking into account fisheries, changes in productivity and trophic relationships. The present document presents briefly the methodology used and the preliminary findings for a few commercially important species.

The ecosystem model account for the biomass of each functional group of species, their diet composition, consumption per unit of biomass (Q/B), mortality from predators and fishing, accumulation of biomass and net migration. The principle behind this ecosystem modelling approach is that, on a yearly basis, biomass and energy in an ecosystem are conserved (Walters *et al.*, 1997). We built a model for the Bay of Biscay and Celtic Sea for 1980, using the Ecopath with Ecosim software (EwE) (Christensen and Walters, 2004).

Of the numerous species that are exploited in the ecosystem only a mere dozen were the subject of stock assessment because of their importance for the industrial fleets while most coastal species were never assessed. The model is articulated around the industrial species: cod (*Gadus morhua*), hake (*Merluccius merluccius*), megrim (*Lepidorhombus whiffiagonis*), anchovy (*Engraulis encrasicolus*), mackerel (*Scomber scombrus*), horse mackerel (*Trachurus trachurus*), whiting (*Merlangius merlangus*), haddock (*Melanogrammus aeglefinus*), plaice (*Pleuronectes platessa*), sole (*Solea solea*), herring (*Clupea harengus*), monkfish (*Lophius piscatorius* and *L. budegassa*) and Norway lobster (*Nephrops norvegicus*), their prey and predators, for a total of 38 groups (see Appendix). Cod, hake and Norway lobster were separated in juvenile and adult stanzas to account for species size-structured interactions among themselves and the fisheries. Juvenile and adult groups are linked and their respective P/B and growth are



1. Study area (shaded).

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calculated from a baseline estimate of the leading group (adults in our case) following a von Bertalanffy growth curve and assuming stable survivorship through ages (Christensen *et al.*, 2005).

Landings, biomass, and fishing mortalities for 1980–2006 were assembled from stock assessment reports and related publications. Landings for other species were taken from the ICES database (<http://www.ices.dk/products/fishstats.asp>), and their biomass trends estimated from the French surveys for the Bay of Biscay (1987–2004, data to be updated). Fish mortality was typically taken from the literature (e.g. stock assessment reports) for commercial species. For other species, natural mortality was derived from an empirical equation based on temperature and growth parameters (Pauly, 1980) or as a function of life expectancy (Hoenig, 1983). The production per unit of biomass (P/B) values result from the sum of natural mortality and fishing mortality. Q/B were estimated from the empirical function based on fish size, diet, and tail size as an index of metabolic rate (Palomares and Pauly, 1989). Diet compositions were obtained from published studies and completed with information from FishBase (Froese and Pauly, 2009) when necessary. The functional predator–prey relationship included in Ecosim is based on the foraging arena theory, which divides the biomass of all prey into vulnerable and invulnerable pools (Walters and Kitchell, 2001). The transfer rate between these two pools (called ‘vulnerability’) can range from one to infinity. Higher rates implying a top-down situation and also means that the predator initial biomass is low compared to its carrying capacity (see Christensen and Walters, 2004; Walters and Martell, 2004).

Starting from 1980 we fitted our model to biomass and landing data using time series of fishing mortality used as an index of fishing effort. Model fitting was achieved by estimating the value of vulnerabilities that would minimize the weighted sum of squares of deviations (SS) between logarithms of observed and predicted biomasses and landings, for all species for which time series were available (Christensen and Walters, 2004).

We also used various climate indices, the North Atlantic Oscillation index (NAO) and sea surface temperature, to modify phytoplankton’s production and obtain better fits. Monthly values of NAO were taken from the Climatic Research Unit web site (<http://www.cru.uea.ac.uk/~timo/datapages/naoi.htm>) and the sea surface temperature (SST) from ICES data base (<http://www.ices.dk/Ocean/data/surface/surface.htm>). Primary productivity and phytoplankton biomass were obtained from Ifremer (Martin Huret, Ifremer, pers. comm.). In addition, we fitted the model by asking Ecosim to estimate both the vulnerabilities and the climate production **anomalies** that would explain the deviations between observed and predicted trends in biomass and landings (see Christensen *et al.*, 2005).

The inclusion of sea surface temperature in the model did not yield very different results from the other scenarios. Forcing the model using phytoplankton biomass led to flatter responses due to the lack of clear trends in the data. Thus, these 2 scenarios are not presented here. Results are shown for 3 scenarios: 1. fisheries only; 2. forcing the primary productivity with NAO index; 3. searching for anomalies. Scenarios were compared on the basis of the resulting fit of biomass and landing trends and the overall sum of squares.

Biomass and catch trophic spectra were built, for the starting and the ending years of the period (i.e. 1980 and 2006) using the ET-Transpose routine (Gascuel *et al.*, 2009) included in the EwE software. Such spectra represent the distribution over trophic levels of the whole ecosystem biomass, or the total catches; they provide a synthetic overview of the ecosystem state and of the major changes occurring during the period. Finally, the impact of fishing was estimated using the EcoTroph model (Gascuel, 2005; Gascuel and Pauly, 2009). Based on equations describing the biomass flow through the food web, this model especially provides an estimate of what would have been the ecosystem biomass (and its distribution over trophic levels), in the absence of fishing.

## RESULTS

### *Ecosim*

Using only fisheries effort, Ecosim predicted the general trend in landing and biomass for several demersal species such as hake, monkfish (figure 2, thick dark line), while large discrepancies occur for other species such as cod, sole, plaice and whiting. Ecosim predicted relatively stable biomass and low landings for pelagic species such as herring, sardine, anchovy and mackerel, suggesting that fishery mortalities are not sufficient to explain biomass trends and that the observed peaks are probably due to environmental

variations rather than population dynamics alone. Ecosim cannot account for the observed decline in mackerel because the high initial biomass is due to the entry of very large cohorts in the population before 1975, the effect of which are still present in the early 1980s. Thus, fishing alone was not sufficient to have provoked the decline. The sum of squares (ss) reached 374 for the fishery alone scenario.

Forcing the primary production with the NAO index did not improve the fit to time series (ss=376) although discrepancies in biomass and/or landing trends were corrected in some cases (e.g. cod, whiting; Figure 2, thin line). Predicted landings for adult cod are closer to the observed, while biomasses are overestimated in the mid-1990s. Predictions for whiting landings and biomass were also improved. Predictions for sole biomass are accurate for the period 1990-2001 although landings were overestimated.

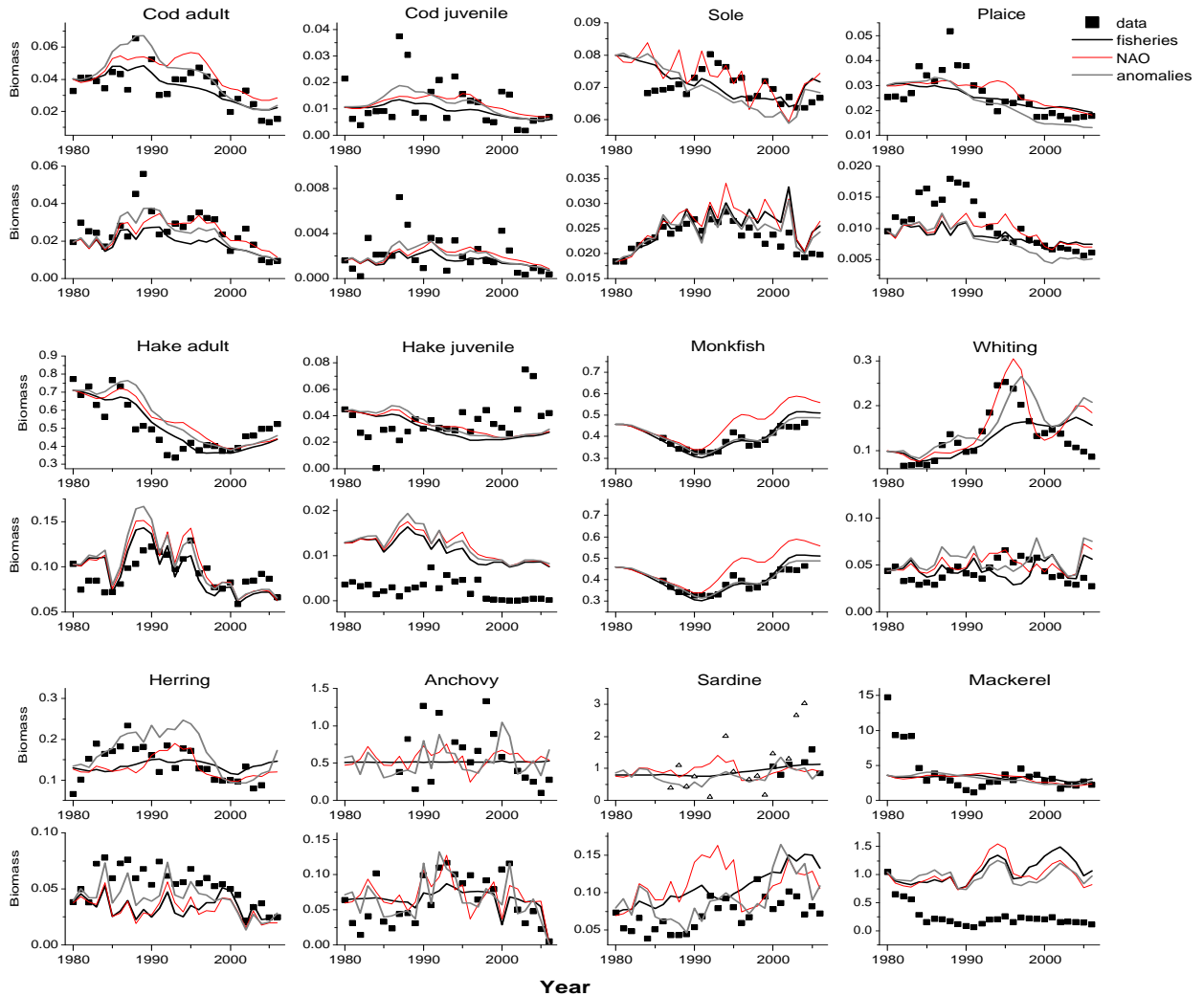


Figure 2 Comparison of biomass ( $t/km^2$ ) and landings ( $t/km^2/an$ ) time series data (squares and triangles) and Ecosim predictions according to 3 scenarios: 1. fisheries only; 2. adding the NAO index; 3. using the anomalies estimated by Ecosim.

However, the trends for pelagics are still poorly predicted although predictions for anchovy landings were better.

The scenario that estimated both climate anomalies and vulnerabilities improved the accuracy of the model general predictions and indeed the sum of squares decreased to 346 (Figure 2, thick gray line). Predictions for adults cod and hake and monkfish (overestimated with the NAO forcing) were closer to the observed trends. Predictions for anchovy and sardine, two species for which biomass trends show large

variation that are not accurate in scale but have improved considerably in trends. The same can be said for herring that shows higher biomass in the period 1985-1995. Still, it has not been possible to predict the large peaks in juvenile biomass for cod in 1987-88 and for hake after 1995.

The NAO index and the anomalies estimated by Ecosim have similar trends (Figure 3B) although the anomalies are shifted in time, reaching maximum positive values ~3-4 years later than the NAO index. This is well illustrated by trend lines estimated with a polynomial function (Figure 3B, thin lines). The resulting phytoplankton biomasses are thus different and especially after 1985 (Figure 3A). The link between each of these indices and the functional groups are still being studied but it seems clear that their respective effect differs among groups. For example, the strong NAO index of the early 1990s seems to correspond to the increased biomass of whiting, sole and sardine).

### EcoTroph

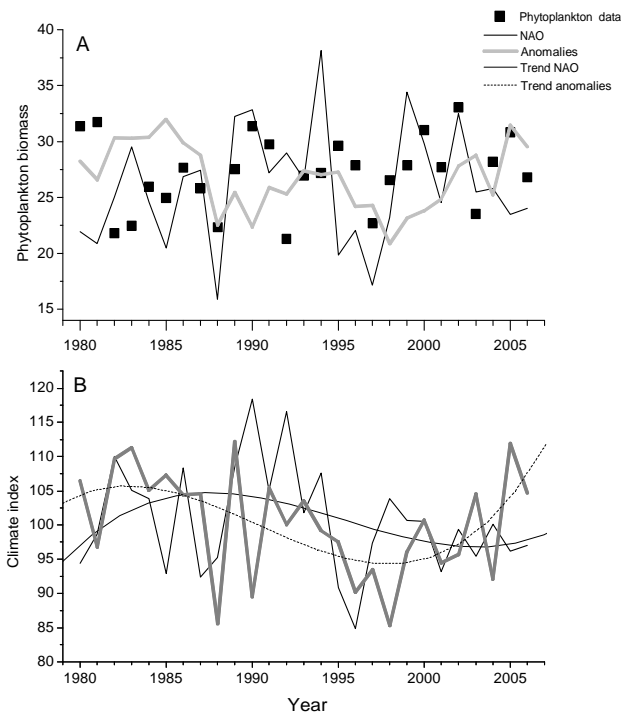
The ecosystem of the Celtic Sea and Bay of Biscay is characterized by a biomass which is relatively abundant for the high trophic levels (more than 1 t/km<sup>2</sup> per trophic class, for instance, for TL = 4.0; see Figure 4 top). The fishery targets mainly these high trophic levels (especially gadoids, monkfish, large pelagics), with fishing mortalities around 0.2 year<sup>-1</sup> for all trophic levels higher than 3.8 (Fig. 4 bottom). Trophic levels between 3.0 and 3.5 (mainly small pelagics, flatfish, crustaceans) are less targeted ( $F < 0.1$  year<sup>-1</sup>). Nevertheless, because they are very abundant, they produce relatively large landings. Finally, the fishery targets a large variety of ecological groups in this ecosystem and landings span a large range of trophic levels, from 2.8 to 4.5 (Fig.4 middle).

Although total landings remain approximately constant between 1980 to 2006, several changes in the trophic structure are noted. First, the biomass of higher trophic levels decreased while it increased for the lower ones. Biomass ratios 2006/1980 show a decrease of about 30 % for trophic levels higher than 4.0 (Figure 5). The same ratio using accessible biomass (i.e. the biomass of exploited groups) suggests that the increase in low trophic levels biomass mainly concerns groups that are not fishable.

Fishing mortalities increased for high trophic level groups, suggesting that the fishermen tried to compensate the decrease in abundance of their traditional stocks. Nevertheless, these increased exploitation rates are not sufficient and the landings of those trophic levels decreased (especially for hake and cod).

In the same period, landings of the lower trophic levels increased. This is due to two processes. On one hand, landings of some lower trophic level species or groups increased (sardine, medium demersals, large crustaceans). On the other hand, the mean trophic level of some groups decreased slightly (according to Ecosim predictions), probably according to a change in the abundance of their preys.

Finally, compared to the unexploited situation, the abundance of the total biomass decreased for all trophic levels higher than 3.5. The rate of decrease exceeds 50 % for trophic levels higher than 4.0 (Fig.5). As a consequence, the mean trophic level of the whole ecosystem biomass decreased from 2.42 to 2.35.



3. Comparison of A. Trends in observed and predicted phytoplankton biomass when using the NAO index and estimated anomalies; B. NAO and anomalies indices and their smoothed trends obtained from a 3rd order polynomial (for illustrative purposes only).

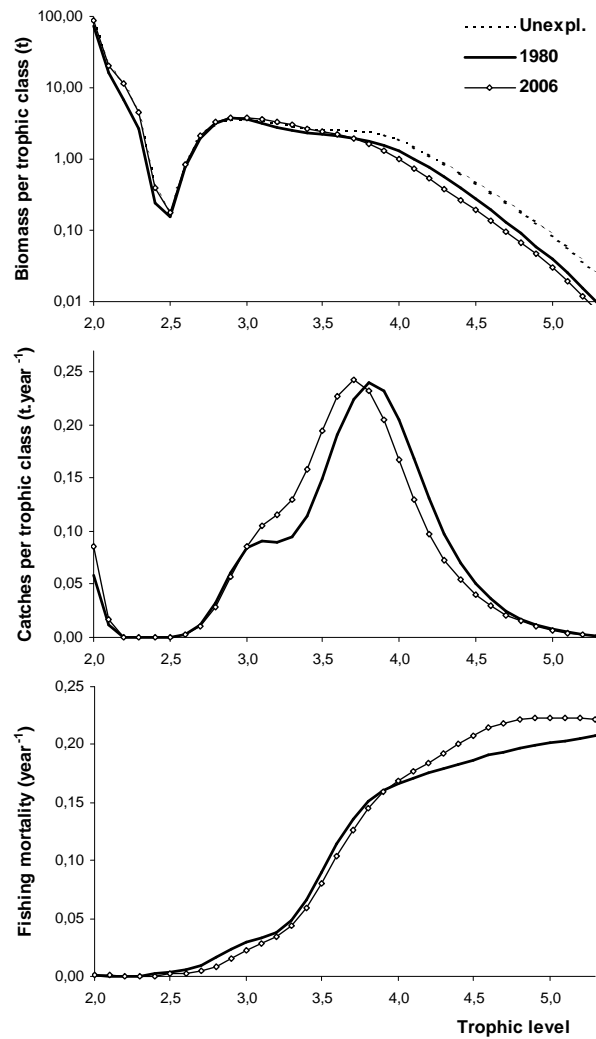


Figure 4. Biomass trophic spectra (top), Landings trophic spectrum (middle) and fishing mortalities per trophic class (down), in the starting year 1980 and the ending year 2006, based on the Ecosim model. Biomass trophic spectrum for the unexploited situation was estimated using the EcoTroph model

## DISCUSSION

The model is still in a preliminary phase but it was still able to predict biomass and catches of most exploited species, from 1980 to 2006. A notable exception to this is mackerel for which we suspect that the main factors influencing its dynamics may be happening outside the study area. Fisheries explain a large part of the trends for demersal species such as cod, hake, and monkfish, while the effect of indices of productivity were necessary to explain a good part of the trends of all fish and more importantly on whiting, sardine, herring and anchovy. It is clear that not all functional groups depend on the same resources, which would explain why the NAO index applies better on some species and the forcing function estimated by Ecosim on others. For example, the juveniles of flatfish are typically found in estuaries where the main influence on survival is likely to be linked to river flow and other factors that are only partly linked to the strength of the NAO index.

Even if biomass only slightly decreases and total landings remain more or less constant during the period, some significant changes occurred. The EcoTroph approach especially shows that the increasing fishing mortality induced a decrease in abundance of predators species. Globally, the ecosystem biomass and landings exhibit decreasing trophic levels, indicating a decrease in functional biodiversity of the underlying ecosystem (Pauly *et al.*, 1998).

Of course, one should not forget that the fishery does not start in 1980. The flat deterioration of the ecosystem health over the last 25 years, is likely to follow a strong and fast deterioration during the decades after World War II, characterized by an huge increase in the European fishing effort (Gu nette *et al.*, in prep.).

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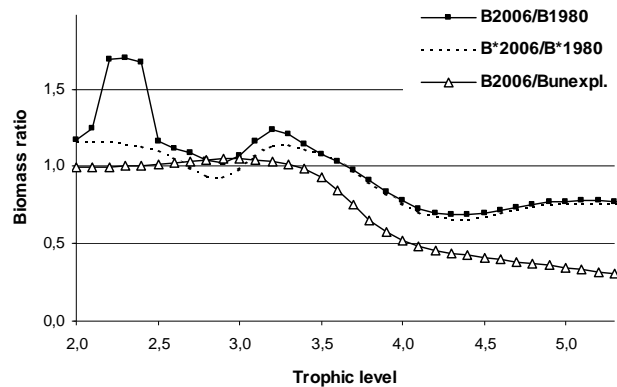


Figure 5. Ratios of biomass per trophic class, between the ending year 2006 and the starting year 1980 or the unexploited situation. B\* refers to the accessible biomass, i.e. the biomass of all species or groups currently fished.

**Appendix 1 - Parameters of the balanced 1980 model. Values in bold have been estimated by Ecopath. The ecotrophic efficiency can be understood as the proportion of the total mortality of the functional group that is explained in the model.**

Group name	Trophic level	Biomass (t/km <sup>2</sup> )	P/B (/year)	Q/B (/year)	Ecotrophic efficiency	P/Q
Baleen whales	3.56	0.063	0.035	4.775	<b>0.17</b>	<b>0.007</b>
Toothed whales	4.34	<b>0.034</b>	0.135	11.998	<b>0.34</b>	<b>0.011</b>
Sharks L	4.47	<b>0.033</b>	0.14	2.265	0.6	<b>0.062</b>
sharks/rays	3.97	<b>0.187</b>	0.52	3.802	0.8	<b>0.137</b>
Whiting	4.40	0.089	1.22	5.708	0.95	<b>0.214</b>
Mackerel	3.76	5.5	0.37	<b>1.48</b>	<b>0.66</b>	0.25
Horse mackerel	3.79	3.2	0.3	<b>1.2</b>	<b>0.94</b>	0.25
Anchovy	3.03	<b>0.412</b>	1.6	9.13	0.95	<b>0.175</b>
Sardine	3.03	0.8	1.2	6.8	<b>0.87</b>	<b>0.176</b>
Sprat	3.14	1	0.73	<b>2.92</b>	<b>0.92</b>	0.25
Herring	3.33	0.15	0.9	4.59	<b>0.99</b>	<b>0.196</b>
Pelagic M	3.24	<b>1.098</b>	0.49	4.92	0.95	<b>0.1</b>
Pelagic L	4.13	<b>0.057</b>	0.43	4.13	0.9	<b>0.104</b>
Hake	4.34	0.64	0.42	2.6	<b>0.54</b>	<b>0.162</b>
Hake juv	4.21	<b>0.0399</b>	0.5	6.38	<b>0.89</b>	<b>0.078</b>
Cod	4.12	0.02	0.96	4	<b>0.75</b>	<b>0.24</b>
Cod juv	3.94	<b>0.0056</b>	0.61	8.097	<b>0.79</b>	<b>0.075</b>
Haddock	3.88	0.045	0.71	3.33	<b>0.86</b>	<b>0.213</b>
Sole	3.16	0.070	0.47	3.7	<b>0.92</b>	<b>0.127</b>
Plaice	3.07	0.020	0.62	3.42	<b>0.97</b>	<b>0.181</b>
Demersal L	3.88	<b>2.303</b>	0.15	<b>0.75</b>	0.85	0.2
Demersal M	3.44	<b>8.073</b>	0.41	<b>2.05</b>	0.95	0.2
Demersal S	3.09	<b>5.424</b>	0.96	<b>3.84</b>	0.95	0.25
Monkfish	4.38	<b>0.292</b>	0.3	2.4	0.9	<b>0.125</b>
Bathy L	4.07	<b>0.415</b>	0.23	<b>1.15</b>	0.9	0.2
Megrim	4.03	0.290	0.49	3.6	<b>0.56</b>	<b>0.136</b>
Bathy M	3.06	0.300	1	5	<b>0.94</b>	<b>0.2</b>
Bathy S	3.06	1.080	1.26	10.97	<b>0.89</b>	<b>0.115</b>
Cephalopods	3.34	<b>0.665</b>	3.68	10	0.9	<b>0.368</b>
Norway lobster	3.21	0.130	0.68	3.4	<b>0.89</b>	<b>0.2</b>
Norway lobster juv	3.19	<b>0.0026</b>	0.6	10.989	<b>0.84</b>	<b>0.055</b>
L crustaceans	3.04	<b>0.252</b>	0.8	<b>5.333</b>	0.95	0.15
Shrimps/crabs	2.82	<b>12.747</b>	1.45	<b>9.667</b>	0.95	0.15
Benthos	2.07	<b>69.451</b>	2.1	<b>14</b>	0.9	0.15
Zooplankton L	2.21	<b>9.896</b>	4.3	16.9	0.9	<b>0.254</b>
Zooplankton S	2.03	4	29	120	<b>0.75</b>	<b>0.242</b>
Phytoplankton	1	27	100	-	<b>0.24</b>	-
Benthic producers	1	<b>11.62</b>	1.15	-	0.2	-
Detritus	1	127	-	-	<b>0.35</b>	-