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Report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO)

12–19 April 2018

San Pedro del Pinatar, Spain

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International Council for
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Executive summary

The 2018 meeting of WGECO was held at the Spanish Institute of Oceanography, San Pedro del Pinatar, Spain from 12–19 April 2018. The meeting was attended by 11 delegates from eight countries and was co-chaired by Stefán Ragnarsson (Iceland) and Jeremy Collie (USA). The work conducted was centred on four Terms of Reference that were made by WGECO in addition to which a list of topics related to pulse trawling was prepared for WGELECTRA.

In recent years, the management of marine fish stocks in European waters is, to an increasing degree, successful in regulating stocks to abundances at or above those producing Maximum Sustainable Yield (MSY). Widespread recovery is expected to lead to increased strength of ecological interactions (predation, competition), both within and between the recovered stocks, and with other components of the ecosystem. Density-dependent growth reduction in response to recovery in plaice in the North Sea and the Eastern Channel, in North Sea dab and in haddock in the Gulf of Maine and winter flounder and haddock on Georges Bank were reported. In Georges Bank haddock, density-dependent growth did not occur in the 1960s, when the stock was at similar abundance as it currently is, which may reflect changes in benthos availability. This work also suggested that the consumption of benthos on the northeast US continental shelf is highly variable among benthivore species, both in space and time. We find limited evidence that density-dependent growth reduction in one stock carries over to other stocks with a high degree of diet overlap. Finally, we have reviewed the potential effect of recovering whale populations on their prey biomass. Some examples from studies and modelling work suggested that this predation can be of such magnitude that it can cause conflicts with fisheries. On the basis of this year’s work, we feel that it is safe to conclude that density-dependent growth reduction is a commonly observed pattern in stocks that show strong recovery. For the coming years, we will focus on understanding when it does (not) occur, and on its consequences for ecosystem and fish stock management.

The impact of fishing on the size structure of demersal fish communities was investigated through an analysis of indicators of Good Environmental Status (total biomass, total catch, size composition and species composition) within demersal fish communities using both survey data and commercial fishing data. A methodology was explored to determine baselines for species composition and a promising empirical approach developed. A preliminary evaluation of observer data from English and Irish fisheries alongside the demersal trawl survey data for the Celtic Sea and Irish Sea demonstrated that the species composition within demersal fish communities and the total biomass of the communities has responded to the impacts of these fisheries. A negative correlation was found between the total commercial catch in the Celtic Sea and the size structure of the surveyed community in the next year (mean maximum length). In the Irish Sea, a great reduction in commercial catch over time was followed by a positive trend in the size structure of the community. The addition of commercial data from other nations fishing in the Celtic Sea should be collated and investigated in future.

WGECO compared different methods used to identify species sensitivity to fishing and found that the relative sensitivity of different species depended on the method used to estimate species sensitivity. While all methods agreed on the high sensitivity of *Dipturus* sp. (flapper skate/blue skate), other species changed in ranking between methods. For example, in the three studies considering spurdog, the first concluded it to be more

sensitive than cod, the second concluded the opposite and the third estimated the sensitivity of the two to be approximately equal. In addition to the definition of sensitive species, WGECO considered indicators to monitor changes in abundances of sensitive species. The group found that the best available information should be used to estimate abundance, and hence where agreed stock assessments exist, these should be preferred to trends in catch rates in individual surveys. Due to the high variance in estimates of abundance for rarer species, longer time periods will be required to detect a change in the mean or a trend (e.g. trends and means need to be estimated based on longer periods than five or one year, respectively).

WGECO investigated a range of possible indicators that could be developed and provided as routine products from ICES. The candidate indicators were Total mortality, Productivity of key predators, Primary production required to support fisheries, Guild level biomass, Total biomass of small fish, Pelagic-to-demersal ratio, and Benthic indicators. These were considered in terms of their value and utility, and of data and methodology availability.1) *Total Mortality* - WGECO considered this to be an indicator which has peer-reviewed data available in some areas, is potentially useful and can be estimated on a routine basis. 2) *Productivity of key predators* - WGECO considered this to be an indicator which has reviewed data available, is potentially useful and can be estimated on a routine basis. 3) *Primary production required to support fisheries* - WGECO considered that the data are not yet available to support the assumptions of specific values of Trophic Transfer Efficiency and also for estimating Trophic level for each species (and variability of that); therefore this indicator is not yet considered operational. 4) *Guild level biomass* - WGECO considered this to be an indicator which has peer-reviewed data available, is potentially useful and can be estimated on a routine basis. 5) *Total biomass of small fish* - WGECO considered that while the indicator is reasonably easy to calculate, it is difficult to see what value it would have, even as a surveillance indicator. 6) *Pelagic-to-demersal ratio* - WGECO considered this to be an indicator which has reviewed data available, is potentially useful and can be estimated on a routine basis. 7) *Benthic indicators* - WGECO did not address benthic indicators in detail as detailed evaluation of their performance has been carried out elsewhere. 8) *Distributional Indicators* - WGECO considered these indicators to have reviewed data and methods available in some cases, that they are potentially useful and that they can be estimated on a routine basis. It was agreed that a further review of spatial indicators, and in some cases testing by WGECO in 2019 would be appropriate. WGECO also addressed methods to integrate indicators. A review of methods and approaches was carried out. It was considered that this was still an area where consensus on approach had not yet been achieved, and that aggregation approaches depended greatly on the needs of the customer. It was concluded that this could be a ToR for WGECO at a later date (possibly 2020).

WGECO was tasked to prepare a list of topics to be considered by WGELECTRA, at its meeting from 17–19th of April, related to a request from the Dutch Ministry of Agriculture, Nature and Food Quality regarding a comparison of the ecological and environmental effects of traditional beam and pulse trawls. WGECO had an extended discussion on this issue and highlighted number of topics that WGELECTRA could consider. These include suggestions of further work, including comparison of the impacts of beam and pulse trawls on benthic communities, examination of the consequences of pulse trawling occurring in areas not fished with beam trawls, further studies on the effects of electricity on various life stages of fish, and the longer-term impacts of repetitive exposure of benthic organisms to electric pulses. WGECO nominated two of its members to participate in the ADGPULSE meeting from 16–18 May.

1 Opening of the meeting

The 2018 meeting of WGECO was held at the Spanish Institute of Oceanography, San Pedro del Pinatar, Spain from 12–19th of 2018. The meeting was attended by 11 delegates from eight countries and was co-chaired by Stefán Ragnarsson (Iceland) and Jeremy Collie (USA). The participants' list is included as Annex 1. The work conducted was centred on four Terms of Reference that were made by WGECO in addition to which a list of topics related to pulse trawling was prepared for WGELECTRA. The meeting agenda is included as Annex 2.

2 Terms of Reference for the 2018 meeting

2017/2/ACOM27 The **Working Group on the Ecosystem Effects of Fishing Activities** (WGECO), chaired by Jeremy Collie, US and Stefan Ragnarsson, Iceland, will meet in San Pedro del Pinatar, Spain 12–19 April 2018 to:

a) Investigate the ecological consequences of stock rebuilding, with particular emphasis on benthivorous fish and invertebrates.
   i) Make first-order estimates of predation pressure on benthos;
   ii) Examine evidence of food limitation and density-dependent growth;
   iii) Compare the footprints of trawling to the footprints of predation pressure on benthos.

b) Use empirical data and available multispecies models to examine how the degree of fisheries balance relates to ecosystem status.
   i) Compare the length composition of total catch (landings and discards) to the length composition in the survey for one region (e.g. Irish Sea);
   ii) Use multispecies models (developed by WGSAM) to identify targets for ecological indicators of state (i.e. status) that relate to an acceptable risk of species diversity loss; and
   iii) Use output of multispecies models to investigate how proposed management strategies affect fisheries balance.

c) Examine individual species abundance trends to improve interpretation of assessment outcomes based on the “abundance of a suite of sensitive fish species” indicator. Apply the sensitive species indicator in additional ICES areas.

d) Investigate and report on potentially valuable ecosystem indicators for which full methodology has yet to be developed, and propose methodologies and data sources. To include inter alia: Total mortality, Productivity of key predators, Primary production required to support fisheries, Guild level biomass, Total biomass of small fish, Pelagic-to-demersal ratio, and Benthic indicators. The current progress in the development of distributional indicators will be reviewed. Furthermore, this ToR should scope and evaluate methods to integrate indicators.

e) Prepare a list of topics to be considered by WGELECTRA (17–19 April 2018) when answering a request from the Dutch Ministry of Agriculture, Nature and Food Quality to compare the ecological and environmental effects of using traditional beam trawls or pulse trawls when exploiting the TAC of North Sea sole, on (i) the sustainable exploitation of the target species (species and size selectivity); (ii) target and non-target species that are exposed to the gear but are not retained (injuries and mortality); (iii) the mechanical disturbance of the seabed; (iv) the structure and functioning of the benthic ecosystem; and to assess (v) the impact of repetitive exposure to the two gear types on marine organisms.

WGECO must report to WGELECTRA on ToR e) before 17 April 2018.

WGECO will report by 3 May 2018 for the attention of the Advisory Committee.

3 ToR a: Investigate the ecological consequences of stock rebuilding, with particular emphasis on benthivorous fish and invertebrates

3.1 General remarks

This is the first year that this has been part of the terms of reference for WGECO, and we expect it to be part of the ToRs for at least two more years. The work presented here reflects that and should be viewed as a first exploration of the subject, and a work in progress.

3.2 Stock recovery

In recent years, the management of marine fish stocks in European waters is, to an increasing degree, successful in regulating stocks to abundances at or above those producing Maximum Sustainable Yield (MSY; Fernandes and Cook, 2013). Widespread recovery is expected to lead to increased strength of ecological interactions (e.g. predation and competition), both within and between the recovered stocks, and with other components of the ecosystem. It has been argued that rebuilding all stocks to their single-species MSY is impossible, when in reality, stocks engage in a multitude of ecological interactions (Larkin, 1977).

3.2.1 Potential stock consequences of recovery

As an increasing number of European fish and shellfish stocks recover, density- or food-dependent effects are expected to occur more frequently. One such effect is reduced individual growth, as an increasing fish biomass ultimately depends on a constant primary production, leaving fewer resources available per capita. This reduced growth has important consequences for fisheries and management, both on the single-stock level and in the context of ecosystem-based fisheries management. Although the Spawning–Stock Biomass (SSB) may rebuild to a very large number, reduced individual growth shifts the population size distribution towards smaller-sized individuals which generally have lower per unit weight market value. In addition, the adults are expected to be in poor physical condition, so that they contain even less edible biomass per unit weight. Slow individual growth also means that a relatively large fraction of total resource intake is 'lost' on metabolic costs, and production of harvestable biomass is relatively inefficient (Ricker, 1975). Lastly, smaller adults, especially those in poor condition, often produce lower quality eggs than larger adults, so that although SSB may be high, recruitment may suffer.

In fact, a general prediction of these stock assessment models is that reduced fishing mortality (through increased life expectancy), leads to larger individuals in the population, which is the exact opposite of what is predicted when density-dependent growth is important (van Kooten *et al*., 2008).

Analytical stock assessment procedures can to some extent account for changes in size-at-age, as the short-term forecasting used to advise quotas is generally based on the size-at-age estimates from recent years. However, this may still lead to errors in the advice during periods of large changes in size-at-age. The problem is likely to be larger for the derivation of reference points, particularly B_{MSY}, which are determined under the assumption that size-at-age is fixed. This problem is illustrated by the basic assumption in these calculations that higher fishing intensity leads to smaller fish, as it reduces the life expectancy. When density-dependent growth reduction is important, this pattern is reversed, as fishing leads to faster growth of the remaining fish (van Kooten *et al.*, 2008).

3.2.2 Potential ecosystem consequences of stock recovery

Any species which depend on the same resources as the rebuilding stock runs the risk of experiencing the same individual growth reduction, even when their own abundance may be low. The shift in a stock's size distribution may also release predation pressure on resources used specifically by large individuals (e.g. fish predation in cod). Furthermore, it has been shown in models, laboratory experiments and field studies (de Roos *et al.*, 2007b; Persson *et al.*, 2007; Schroder *et al.*, 2014) that in populations which experience density-dependent growth, the effect of size-dependent mortality (as imposed by most fisheries) can be counterintuitive. In the simplest example, thinning of small individuals leads to improved growth of survivors and a population size distribution with more large individuals (van Kooten *et al.*, 2007). This finding has important ramifications for the role of undersized bycatch and predation mortality in fisheries. However, it can also result in more complex effects of size-dependent mortality, including a positive effect of mortality on the population density of the harvested size range (i.e. fishing mortality leading to more fish in the sea; de Roos *et al.*, 2007a). In a foodweb context, it can induce alternative stable states in predators (and the associated risk of catastrophic collapse; van Kooten *et al.*, 2005; Van Leeuwen *et al.*, 2008; van Denderen and van Kooten, 2013), but also facilitation between competing predators (De Roos *et al.*, 2008).

3.3 North Sea plaice

3.3.1 Density-dependent growth reduction during stock recovery

Over the past decade, the plaice stock in the North Sea has increased to an unprecedented size. During this period, there has also been a reduction in the size-at-age, especially of the older individuals (Figure 3.1). This reduction is so pronounced that the average size of a 6-year-old plaice in 2016 is similar to that of a 3-year-old plaice 20 years ago.

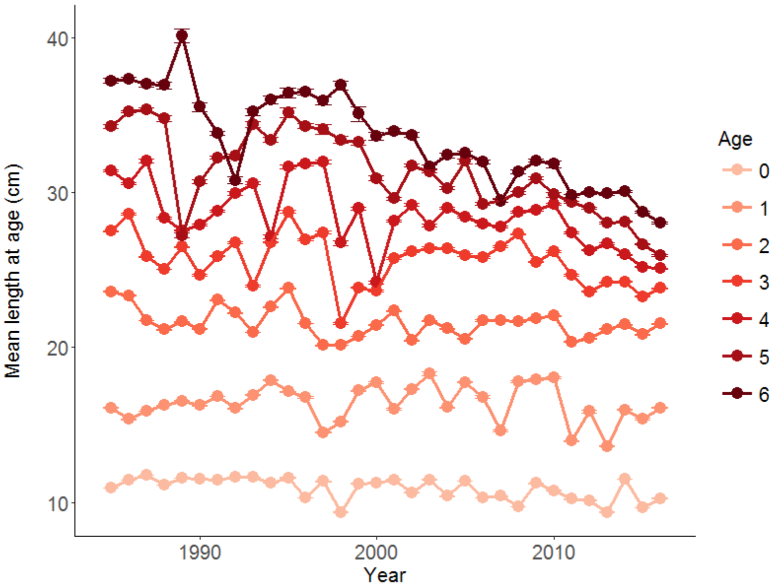

Figure 3.1. Average (±SE) size-at-age of North Sea plaice in the annual third-quarter BTS survey.

The co-occurrence of reduced size-at-age and increasing stock abundance has led to a negative relationship between the two for the recent period (Figure 3.2). This correlative indication of density-dependent growth reduction, is further strengthened by a coinciding reduction in physical condition across a range of sizes (Figure 3.3), hinting that food scarcity may indeed be the mechanism behind the patterns. Further evidence comes from the relationship between the population feeding capacity (expressed as the cpue of squared individual lengths; Kooijman, 2009; Figure 3.4) and the total macrobenthos abundance as measured in the Dutch national MWTL survey (van Denderen *et al.*, 2014).

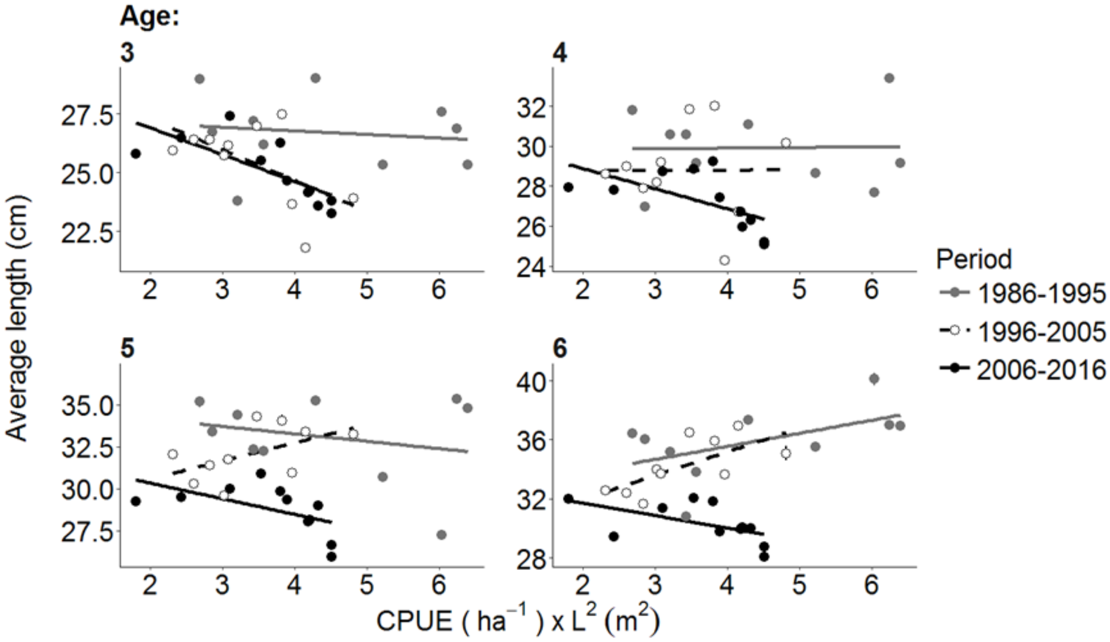

Figure 3.2. Relationships between length-at-age and catch per unit of effort (cpue) of 3, 4, 5 and 6 year old plaice in the North Sea. Cpue is expressed as the sum of the squared length of each plaice in the catch, as intake capacity is generally considered to scale with length squared rather than biomass (Kooijman, 2009). Lines indicate significant linear regressions, points are mean lengths for individuals in each year, ±SE.

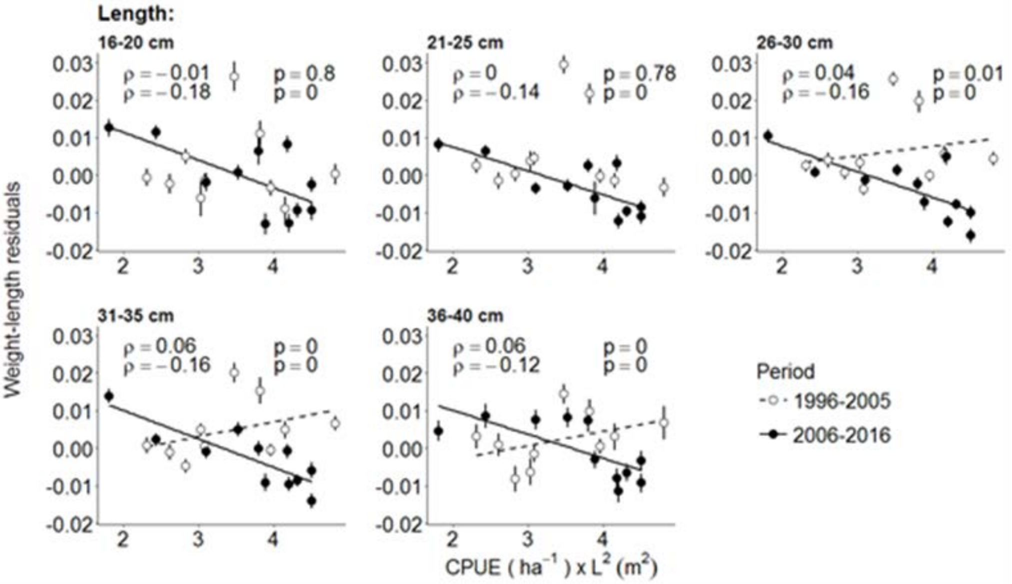

Figure 3.3. Residuals of a fitted length–weight relationship (of the form W=aL^b) of plaice in different size classes caught in the quarter 3 BTS survey in the North Sea. Cpue is expressed as the sum of the squared length of each plaice in the catch, as intake capacity is generally considered to scale with length squared rather than biomass (Kooijman, 2009). Lines indicate significant linear regressions, points are mean residuals for individuals in each year, ±SE.

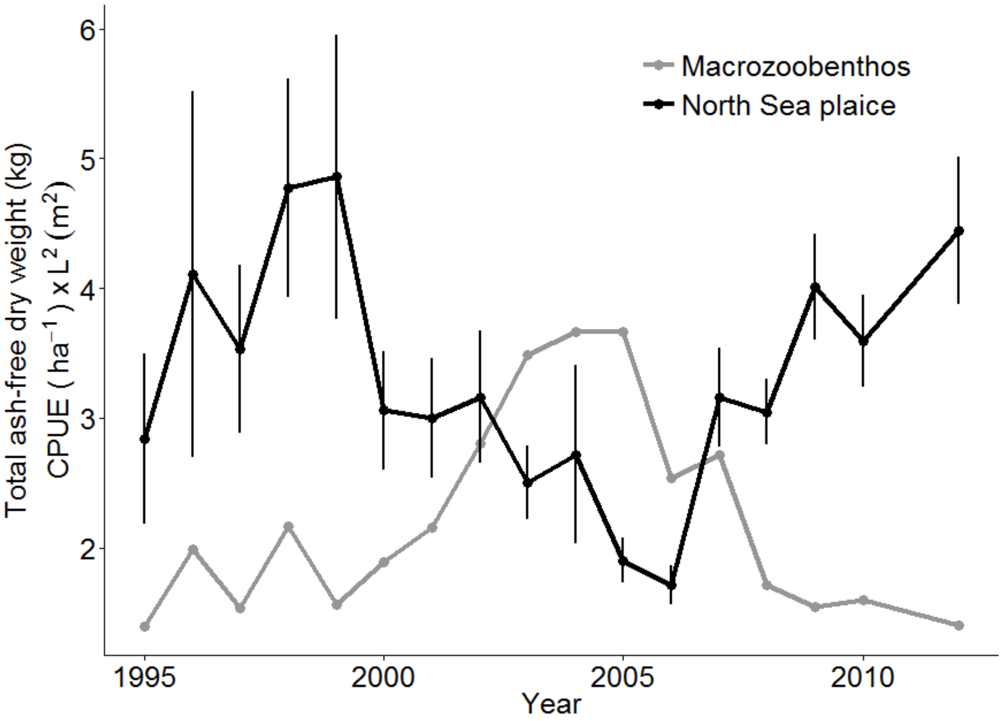

Figure 3.4. Time-trend of macrozoobenthos total ash-free dry weight (kg) in the Dutch MWTL boxcore survey (see van Denderen *et al.*, 2014 for details) and the feeding capacity of the North Sea plaice population as sampled in the 3rd quarter BTS survey (individuals over 15 cm, expressed as the total squared length of all individuals caught).

Density-dependent individual growth in harvested fish is generally not expected in stocks for which fishing mortality strongly depresses abundance. Indeed, an earlier analysis using data up to 2000, before the rebuilding of the stock, indicated no density-dependent growth in North Sea plaice (Lorenzen and Enberg, 2002). More recently, an analysis using data up to 2014 reported patterns similar to those presented here (van der Sleen *et al*., 2018). Another recent paper, which analysed the relative strength of density-dependent recruitment and growth also found significant negative density-dependent growth in North Sea plaice (Zimmerman *et al*., 2018).

3.4 Patterns of density-dependent growth in plaice year classes

Several studies have looked at the relationship between growth or size of plaice and its abundance. Lorenzen and Enberg (2002) concluded that density-dependent growth did not occur in North Sea plaice, but they used data only up to 2000, before the recent population increase. More recent analyses that include later years do find clear evidence of density-dependent growth reduction (van der Sleen *et al*., 2018; Zimmerman *et al*., 2018). This suggests that the phenomenon started to occur during the rebuilding of the stock. However, even when SSB is low, there is a large variation between years in recruitment, with an exceptionally large or small year class appearing occasionally. In absence of density-dependent growth, there should be no relationship between year-class biomass and size, but this is not the case (Figure 3.5). We find a clear and negative relationship between year-class size (relative to the long-term mean year-class size) and the weight of the individuals in the year class (relative to long-term mean individual weight). We also find a relative absence of datapoints in the top right plane of the figure, indicating a strong underrepresentation of year classes that are both numerous and consist of large individuals. This pattern is similar for all ages 1–10+ and the relationship is significant ($p<0.01$) for all ages 2–10+ (results not shown). This result clearly shows that although density-dependent growth may not be apparent at the population level (SSB) under high fishing pressure, it is still an important process limiting the growth of large year classes.

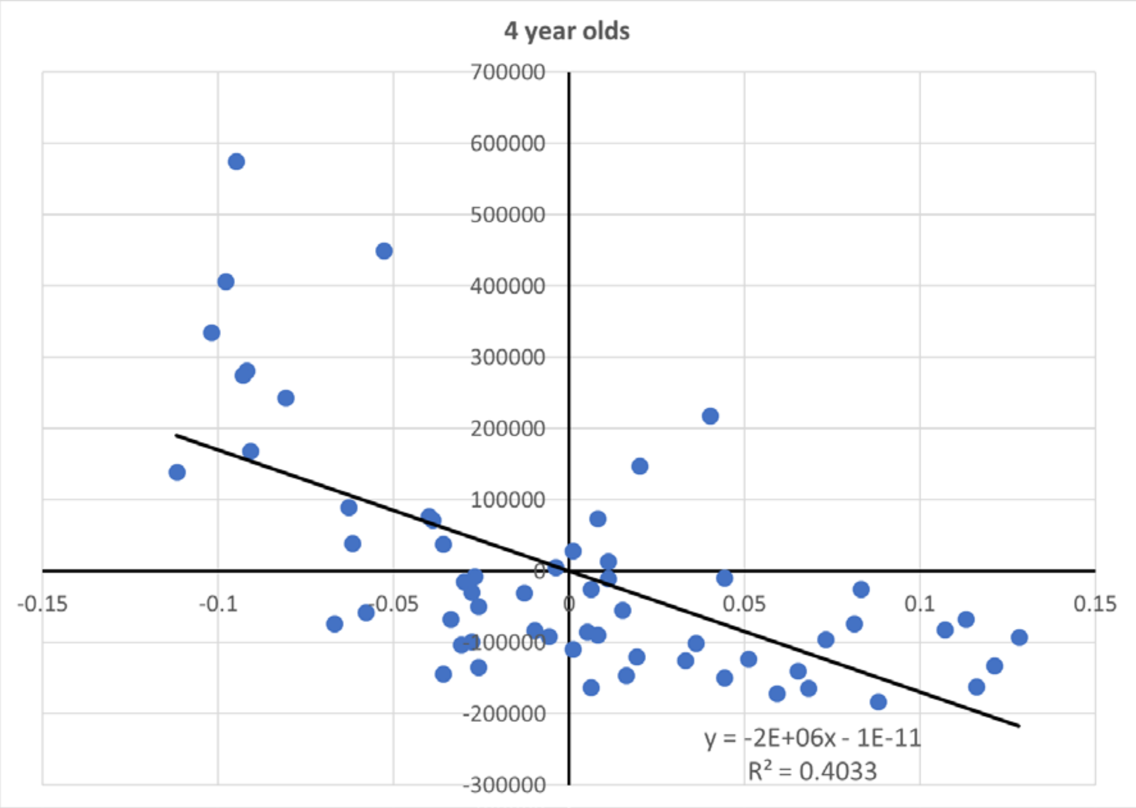

Figure 3.5. Individual weight (kg, x-axis) vs. number of individuals of age 4, relative to the long-term mean, based on stock assessment data 1957–2017 (ICES 2017a).

3.4.1 Growth of North Sea dab

Dab (*Limanda limanda*) is generally considered to be an ecologically similar species to plaice. However, significant differences exist between dab and plaice in diet preference (Johnson *et al.*, 2015). Although there is no analytical assessment for North Sea dab, survey indices indicate that it has also increased strongly in recent years (Figure 3.6). This increase is likely the result of reduced fishing mortality on plaice and sole, as they are generally caught in the same fishery. The patterns observed in dab length-at-age (Figure 3.7) and condition (not shown) are highly similar to those in plaice. The fact that dab have a distinct diet from plaice implies that the similarities in the growth patterns are more likely to result from concurring self-limitation than from interspecific competition between dab and plaice.

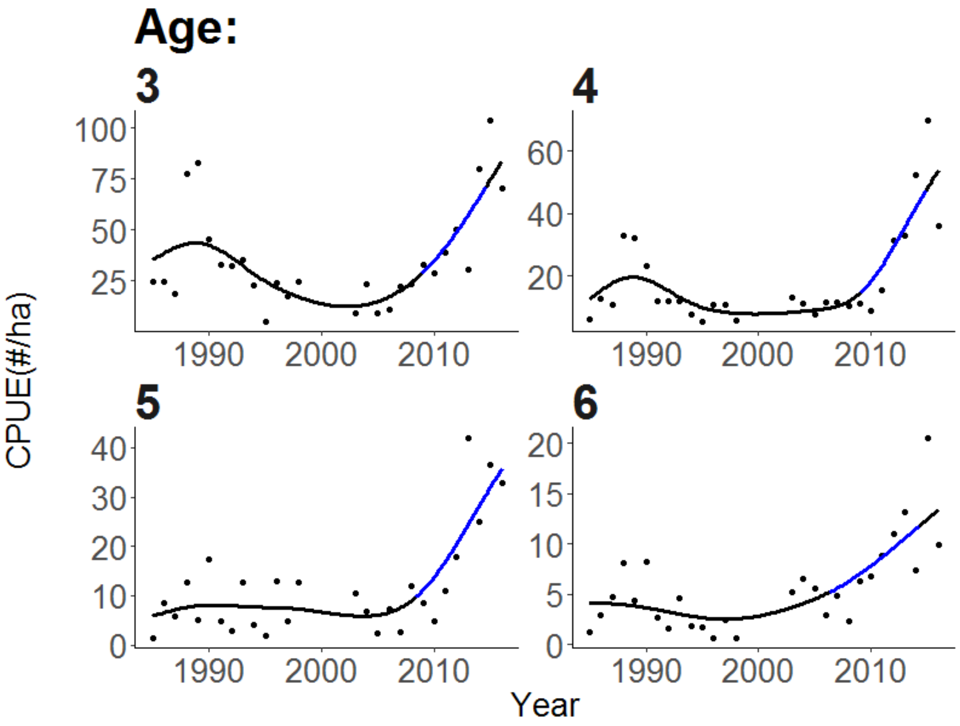

Figure 3.6. Dab catch per unit of effort (Numbers per hectare) by age in the BTS survey over time.

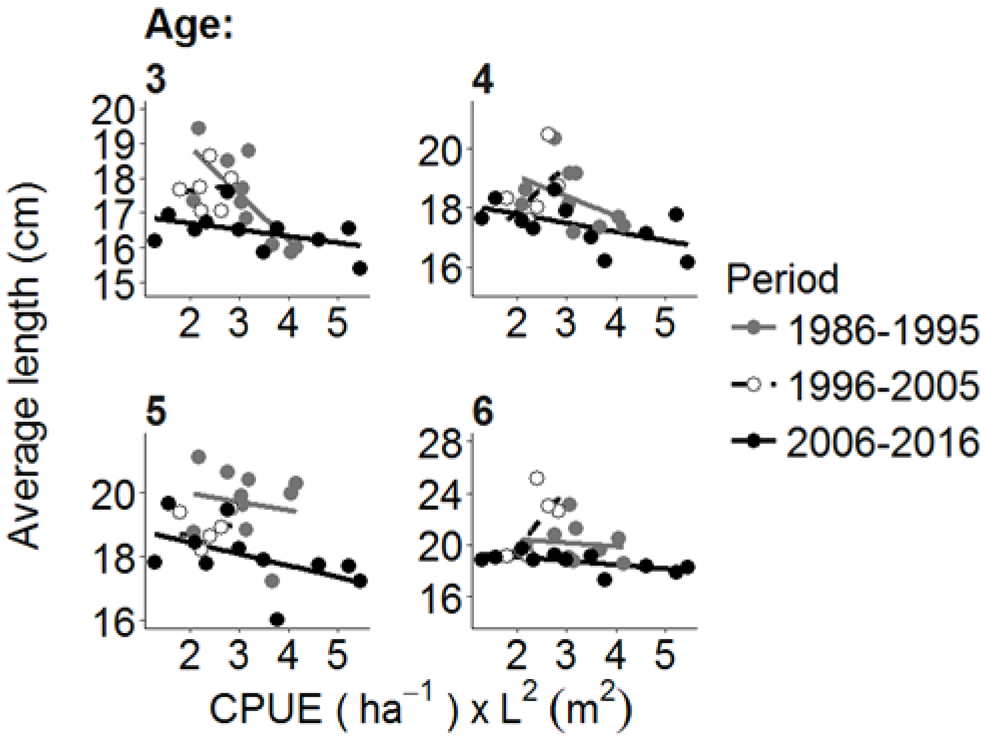

Figure 3.7. Relationships between length-at-age and catch per unit of effort (cpue) of 3, 4, 5 and 6-year-old dab in the North Sea. Cpue is expressed as the sum of the squared length of each dab in the catch, as intake capacity is generally considered to scale with length squared rather than biomass (Kooijman, 2009). Lines indicate significant linear regressions; points are mean lengths for individuals in each year.

3.5 Patterns in other plaice populations

3.5.1 ICES perception of plaice stocks across European waters

Here we extend part of the North Sea plaice analysis to other plaice stocks in the ICES area. Many (but not all) of these stocks have seen similar recovery as the North Sea stock in the last decade (Figure 3.8). By repeating the analysis for other stocks of the same species, we aim to get a better understanding of the factors determining the occurrence of density-dependent growth at high density.

For the Celtic Sea South, southwest of Ireland, "the apparent reduction in SSB since the early 2000s is mainly driven by a reduction in abundance of young fish in recent years and high fishing mortality. However, it is unclear whether this lack of young fish in the landings (and commercial tuning lpue index) is due to increased discarding or poor recruitment." (ICES, 2017b).

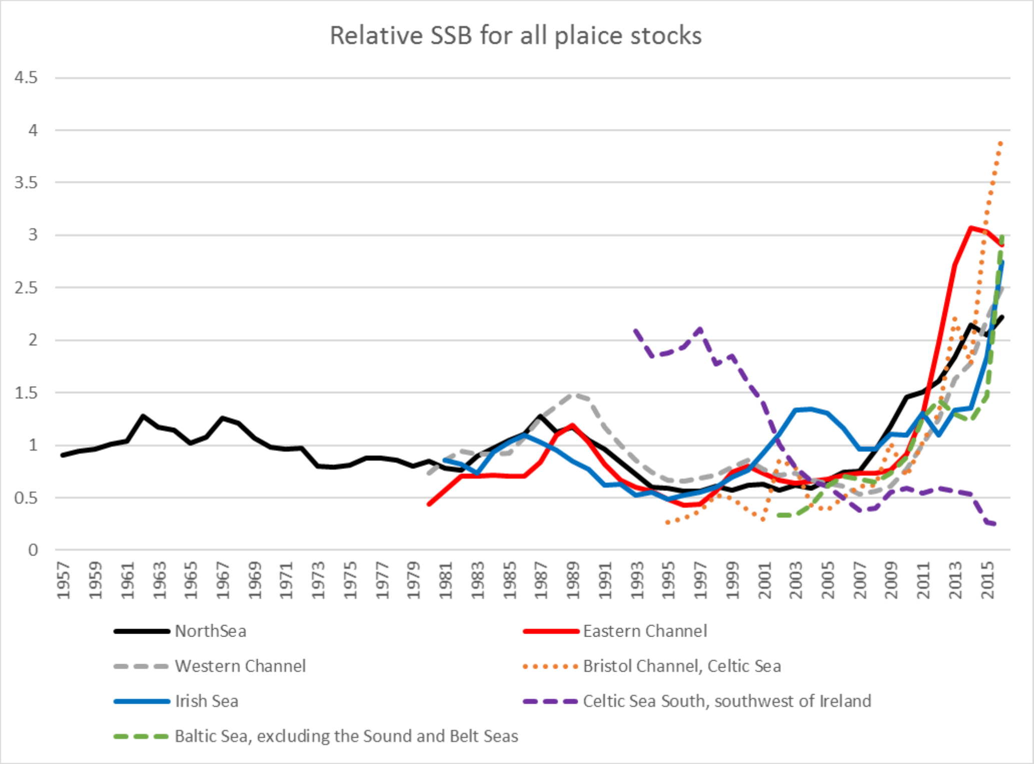

Figure 3.8. Evolution of the relative SBB (normalized by time-series average) for plaice stocks in European waters. Solid lines refer to category 1 stock assessment (absolute SSB), broken lines to category 3 stock assessments (trend-based), and dotted line to survey trends (Bristol Channel, Celtic Sea: UK(E&W)-BTS-Q3 (kg/km)).

3.5.2 Exploration of the weight-at-age patterns in Plaice stocks of the Eastern Channel (7.d), Celtic Sea (7.fg) and Irish Sea (7.a), based on the UK-BTS survey

Weight-at-age patterns were explored for three of these stocks using the UK BTS-Q3 survey (Figure 3.9). In a first approach, total cpue (all ages) was used as a proxy of population size and thus consumer abundance. We find a clear and negative relationship between the average individual weight per age class and year, and the total cpue per year in the Eastern English Channel area stock, which is adjacent to the North Sea stock, and the second stock in estimated SSB after the North Sea stock (Figure 3.10). In the other two regions, i.e. the Bristol Channel/Celtic Sea (7.fg) and the Irish Sea (7.a), no significant relationships were found (results not shown). It is worth noting that these stocks are smaller than the 7.d stock (Estimated 2016 SSB for the Irish Sea 22 686 tvs.68 985 t for the English Eastern Channel). These data will be further explored and similar analysis will be carried out on the other plaice stocks (for which survey data are available).

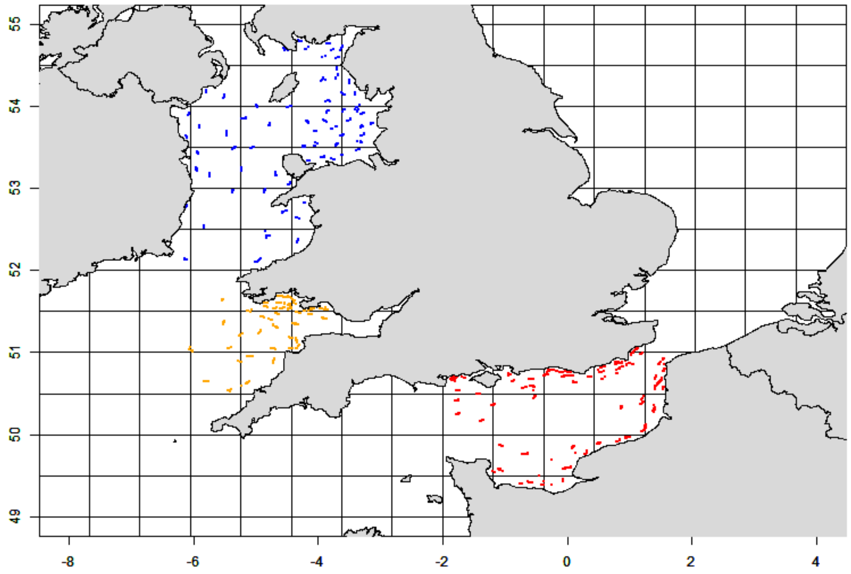

Figure 3.9. Distribution of the UK-BTS hauls (2008–2017) in the three areas investigated: 7.a (blue), 7.fg (orange), and 7.d (red).

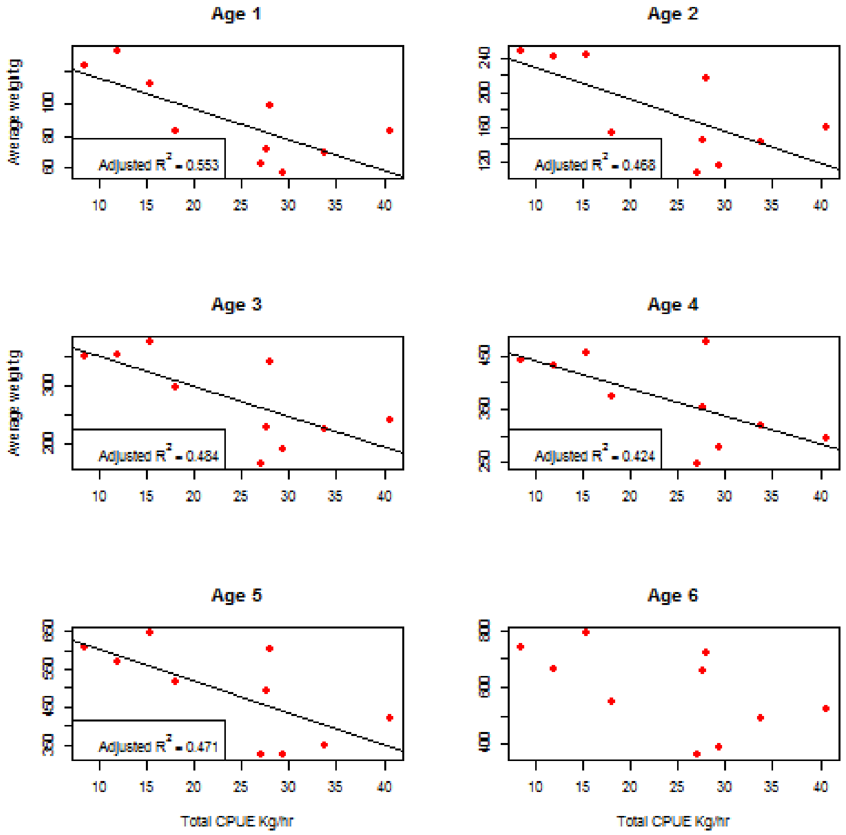

Figure 3.10. Average individual weight (g, y-axis) vs. total cpue (x-axis) for ages 1 to 6 in the Eastern English Channel. Lines indicate significant linear regression.

3.6 First-order estimates of predation pressure on benthos, and examining food limitation and density-dependent growth of fishes of the Northeast US continental shelf

3.6.1 Introduction

Fish diet data from NOAA Fisheries, Northeast Fisheries Science Center (NEFSC) bottom-trawl surveys were evaluated for 14 benthic invertebrate fish predators of the Northeast US continental shelf (Table 3.1). The total amount of food eaten and the type of food eaten, were the primary data examined. From these basic diet data, composition of benthic invertebrates, per-capita consumption, and the total amount of benthic invertebrates (herein termed benthos) removed by six of the 14 predators by stock area were calculated (Table 3.2).

Estimates of total amount of benthos consumed relative to the biomass of benthos (g m^2) on Georges Bank was also considered for five of the 14 predators. This provided a total percent of benthic biomass consumed by these fishes.

Weight-at-age data from stock assessments were obtained to examine density-dependent growth and its relationship with feeding and food limitation.

Table 3.1. Fourteen benthivores from the NEFSC fish diet database.

Species	Common name
Hippoglossoides platessoides	American plaice
Centropristis striata	Black sea bass
Gadus morhua	Cod
Melanogrammus aeglefinus	Haddock
Leucoraja erinacea	Little skate
Myoxocephalus octodecemspinosus	Longhorn sculpin
Zoarces americanus	Ocean pout
Urophycis chuss	Red hake
Mustelus canis	Smooth dogfish
Amblyraja radiata	Thorny skate
Pseudopleuronectes americanus	Winter flounder
Leucoraja ocellata	Winter skate
Glyptocephalus cynoglossus	Witch flounder
Limanda ferruginea	Yellowtail flounder

Table 3.2. Subset of benthivores and stock areas.

Species	Common name	Area	Density-dependent growth?
Limanda ferruginea	Yellowtail flounder	Gulf of Maine	No
Limanda ferruginea	Yellowtail flounder	SNE—MA	No
Pseudopleuronectes americanus	Winter flounder	Georges Bank	Weak
Pesudopleuronectes americanus	Winter flounder	SNE—MA	No
Hippoglossoides platessoides	American plaice	GoM—GB	No
Glyptocephalus cynoglossus	Witch founder	Unit stock	No
Melanogrammus aeglefinus	Haddock	Gulf of Maine	Yes
Melanogrammus aeglefinus	Haddock	Georges Bank	Yes
Gadus morhua	Atlantic cod	Gulf of Maine	No
Gadus morhua	Atlantic cod	Georges Bank	No

3.6.2 Methods

Every fish predator that contained benthic organisms (namely, polychaetes, gammarids, anthozoans, echinoderms, bivalves, cancer crabs, other crabs, cumaceans, gastropods, holothuridians, and hydrozoans) was identified. From that original list, a subset of the top predators comprising 76% of the occurrences of all benthos predation, included within a benthivorous trophic guild (Garrison and Link, 2000), and having diet information available for at least half of the time-series (1977–2016) were included for estimating benthos consumption. Diet data were spatially aggregated in two ways: 1) For all predators, data were aggregated across the entire Northeast US continental shelf and 2) For American plaice, cod, haddock, winter flounder, witch flounder, and yellowtail flounder, data were also aggregated by stock area (i.e. Gulf of Maine, Georges Bank or Southern New England and Mid-Atlantic Bight; Table 3.2).

Estimates were calculated on a seasonal basis (two 6-month periods) for each predator and summed for each year. Although diet data collections for these predators started quantitatively in 1973 (mainly Order Gadiformes) and extend to the present (through 2016), not all benthivores were sampled during the full extent of this sampling program. Stomach sampling for the non-Gadiformes considered here began in 1977 and extends through 2016; thus, the time-series was restricted to 1977–2016 to consider time-series trends. For more details on the food habits sampling protocols and approaches, see Link and Almeida (2000) and Smith and Link (2010). This sampling programme was part of the NEFSC bottom-trawl survey program; further details of the survey programme can be found in Azarovitz (1981), NEFC (1981), and Reid *et al.* (1999).

Basic diet data

Mean amounts of benthos eaten ($D_{i,t,st}$; as observed from diet sampling) for each predator (*i*), temporal (*t*, fall or spring; year) and in some cases each stock (*st*, e.g. Georges Bank) were weighted by the number of fish at length per tow and the total number of fish per tow as part of a two-stage cluster design (See Link and Almeida, 2000; Latour *et al.*, 2007). These means included empty stomachs, and units for these estimates are in grammes (g).

Numbers of stomachs

The adequacy of stomach sample sizes was assessed with trophic diversity curves by estimating the mean cumulative Shannon–Wiener diversity of stomach contents plotted as a function of stomach number. The order of stomachs sampled was randomized 100 times, and the criteria for asymptotic diversity was met when the slope of the three proceeding mean cumulative values was ≤0.1 which was similar to previous fish trophic studies (e.g. Koen Alonso *et al.*, 2002; Belleggia *et al.*, 2008). A minimum sample size approximately equal to 20 stomachs for each predator per season-year or stock area-season-year emerged as the general cut off for these asymptotes. Annual estimates of diet compositions of benthos were estimated for each predator and season, and considered stock area where applicable. For all species, mean amounts of benthos consumed ($D_{i,t,st}$) were not averaged between years with zero stomachs containing benthos.

Consumption rates

To estimate per-capita consumption, the gastric evacuation rate method was used (Eggers, 1977; Elliott and Persson, 1978). There are several approaches for estimating consumption, but this approach was chosen as it was not overly simplistic (as compared to % body weight; Bajkov, 1935) or overly complex (as compared to highly parameterized bioenergetics models; Kitchell *et al.*, 1977). Additionally, there has been extensive use of these models (Durbin *et al.*, 1983; Ursin *et al.*, 1985; Pennington, 1985; Overholtz *et al.*, 1999, 2000; Tsou and Collie, 2001a, 2001b; Link and Garrison, 2002; Link *et al.*, 2002; Overholtz and Link, 2007; Smith *et al.*, 2016). Units are in g year^{-1}.

Using the evacuation rate model to calculate consumption requires two variables and two parameters. The daily per capita consumption rate of benthos, $C_{i,t,st}$ is calculated as:

$$C_{i,t,st} = 24 \cdot E_{i,t,st} \cdot D_{i,t,st},$$

where 24 is the number of hours in a day. The evacuation rate $E_{i,t,st}$ is:

$$E_{i,t,st} = \alpha e^{\beta T_{i,t,st}},$$

and is formulated such that estimates of mean benthos eaten ($D_{i,t,st}$) and ambient temperature ($T_{i,s,st}$) as stratified mean bottom temperature associated with the presence of each predator from the NEFSC bottom-trawl surveys (Taylor and Bascuñán, 2000; Taylor *et al.*, 2005) are the only data required. The parameters α and β were set as 0.04 and 0.115 and chosen from the literature (Durbin *et al.*, 1983; Tsou and Collie, 2001a; 2001b; Methratta and Link, 2012).

Fish predator abundance estimation

The scaling of total consumption requires information on predator population abundance. Abundance estimates by stock from assessment models were available for recently assessed stocks of the Northeast US continental shelf or swept-area abundance for each predator (Table 4.2). Species estimated with swept-area abundance (Gulf of Maine haddock, Southern New England-Mid-Atlantic Bight winter flounder, and southern New England yellowtail flounder) used an assumed catchability coefficient, $q = 1.0$. If abundance data were not readily available, per-capita consumption estimates were not expanded to a population level.

Scaling consumption

Following the estimation of consumption rates for each predator and temporal (t) scheme or for each predator, temporal (t) and stock (st) scheme, those estimates were scaled up to a seasonal estimate ($C'_{i,t,st}$) by multiplying the number of days in each half year:

$$C'_{i,t,st} = C_{i,t,st} \cdot 182.5.$$

These were then summed to provide an annual estimate, $C'_{i,st,year}$:

$$C'_{i,st,year} = C_{i,st,fall} + C_{i,st,spring}.$$

and were then scaled by the annual stock abundance ($N_{i,st,year}$) to estimate a total annual amount of benthos removed by predator and stock (where considered), $C_{i,st,year}$:

$$C_{i,st,year} = C'_{i,st,year} \cdot N_{i,st,year};$$

The final benthos consumption time-series was 1977–2016. The total consumption of benthos per predator is presented as thousands of metric tonnes year^{-1}.

Georges Bank benthos consumption

Total consumption of benthos by the five predators with stocks on Georges Bank were scaled by the area of this region to provide estimates of total consumption per area (g m^{-2}; Table 3.2). Steele *et al.* (2007) estimated total benthic biomass on Georges Bank to be approximately 70.00 g m^{-2}. With this static value of total benthic biomass, a time-series of percent of total benthos consumed by these predators was estimated.

Growth of benthic feeding fish

A group of species was selected for which weight-at-age data and consumption estimates were available (Table 3.2). The weight-at-age data corresponding to the spawning–stock biomass were extracted from the stock assessment input files. Weight-at-age data were standardized by dividing by the mean weight of each age over the time-series.

A model for weight-at-age has been developed based on energetic principles (Horbowy and Luzenczyk, 2017):

$$W_{a,t} = \frac{aPrey_t}{bPrey_t + Density_t} + \varepsilon_{a,t}$$

Where $W_{a,t}$ is the standardized weight-at-age *a* in year *t*, Prey is prey abundance, Density is predator density and is ε an additive error. This equation describes weight as a hyperbolic function of both Prey and Density. Note that in this formulation we assume that values of the parameters *a* and *b* are shared by all ages. The weight-at-age model was fit to the observations with R function optim.

In this application we used Spawning–Stock Biomass (SSB) as the measure of Density. Other abundance measures could be chosen, but they are likely to be strongly correlated with SSB. The mean weight (kg) of benthic prey in the predator stomachs was used as a proxy for prey abundance. The diet data came from the NEFSC fish diet database (see above).

3.6.3 Results and conclusions

Time-series of consumption

Annual estimates of per capita consumption by predator and season for the 14 benthivores are shown in Figure 3.11. Across the entire Northeast US continental shelf and from 1977–2016, these benthivores consume approximately 0.2 to 2.5 kg of benthos per capita each year. An exception to this was smooth dogfish with an average annual per capita consumption of benthos greater than 30 kg. It was of interest that the per-capita consumption of benthos was not constant over time as multiple benthivores (e.g. cod, haddock, little skate, ocean pout, red hake, thorny skate, and winter skate) showed higher consumption prior to 2000 with a general decrease thereafter (Figure 3.11).

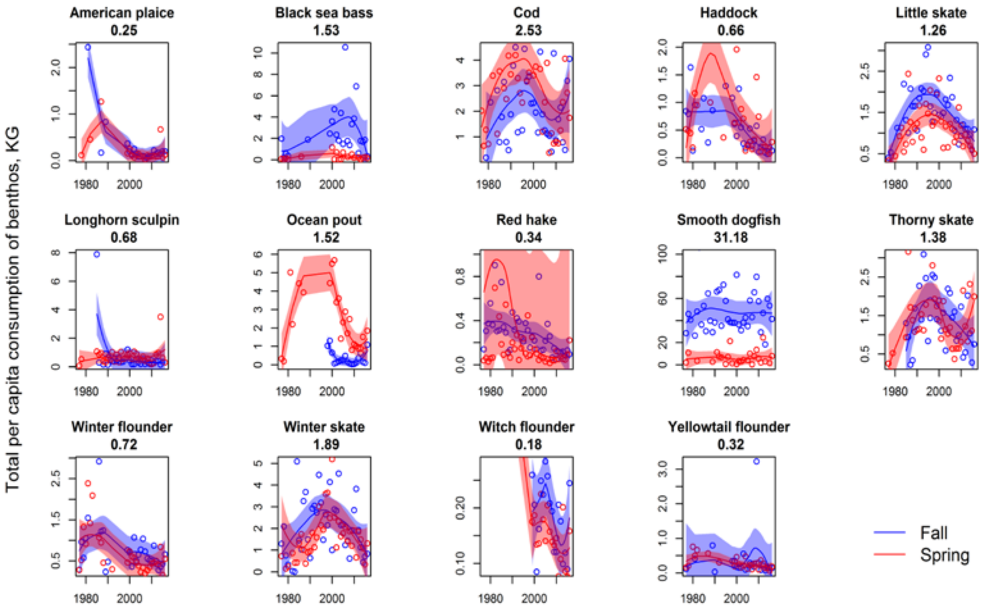

Figure 3.11. Annual time-series of per-capita consumption of benthos (kg) by predator and season for the Northeast US continental shelf. Values under the predator names indicate the average consumption across all years and seasons. Smoother is LOESS with 95% ci, span =0.8.

By stock area, average annual per capita consumption per predator and stock were mostly within the range of shelf-wide estimates (Figure 3.12). The exception was Georges Bank cod with nearly 7 kg per capita consumption of benthos. Time-series trends by predator and stock were similar to trends in consumption for the entire shelf for those predators considered. When scaled to the population level with predator stock abundance, average annual estimates of benthos consumed were on the order of 10s to 100s of thousands of metric tonnes per predator year^{-1} (Figure 3.13). The two flounder species, winter flounder of southern New England and Mid-Atlantic Bight, and yellowtail flounder of southern New England had average annual consumption estimates an order to two orders of magnitude less than the other predators. Interestingly, comparing consumption of these benthivores between Georges Bank and Gulf of Maine revealed that the average annual consumption was generally one to two orders of magnitude greater on Georges Bank. For many of these benthivores, scaled benthos consumption decreased over time, but for some, an increase in consumption was observed in recent years (i.e. Georges Bank and Gulf of Maine haddock; Figure 3.13).

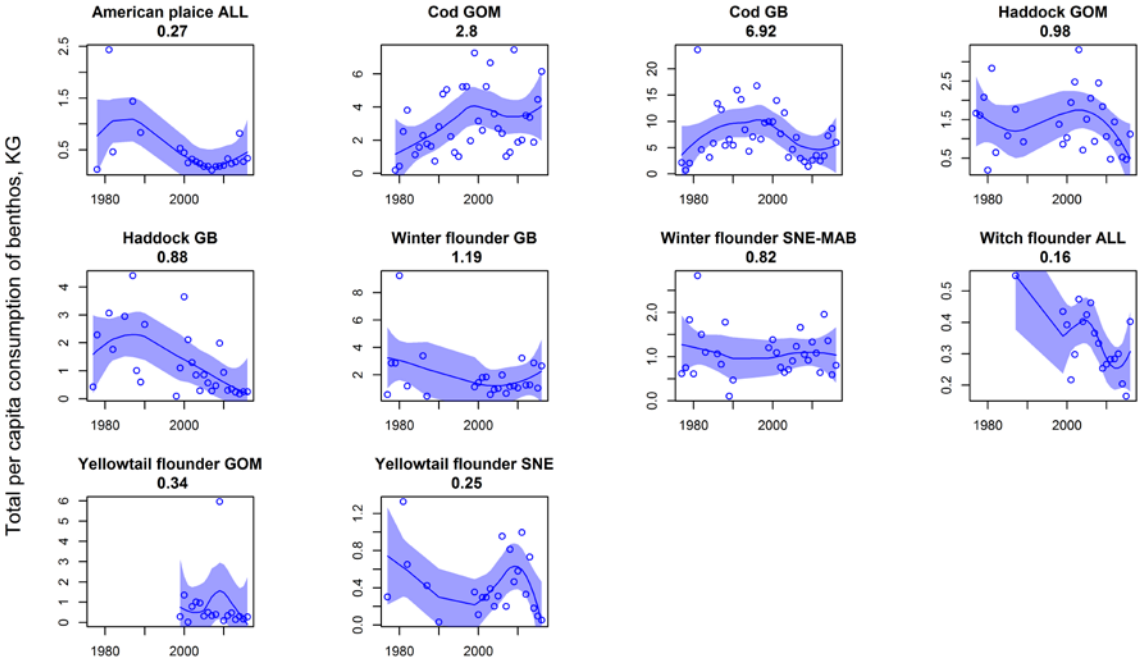

Figure 3.12. Annual time-series of per capita consumption of benthos (kg) by predator and stock area. Values under the predator names indicate the average consumption across all years. Smoother is LOESS with 95% ci, span =0.8. Area abbreviations are GOM (Gulf of Maine), GB (Georges Bank) and SNE-MAB (Southern New England, Mid-Atlantic Bight).

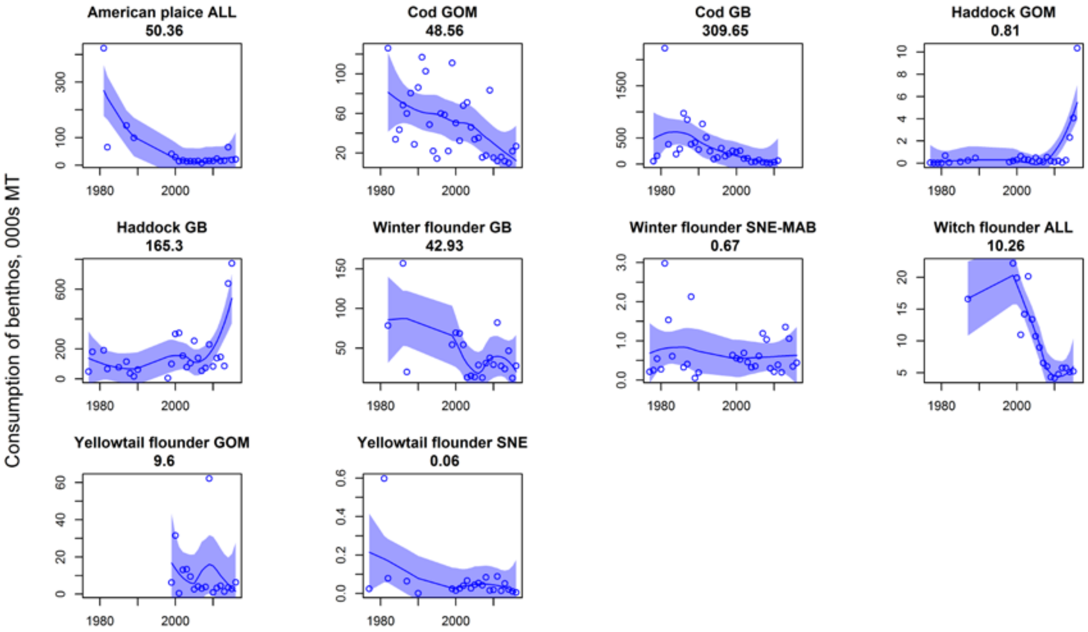

Figure 3.13. Annual time-series of consumption of benthos (000s MT) scaled by predator abundance by predator and stock area. Values under the predator names indicate the average consumption across all years. Smoother is LOESS with 95% ci, span =0.8. Area abbreviations are GOM (Gulf of Maine), GB (Georges Bank) and SNE-MAB (Southern New England, Mid-Atlantic Bight).

Georges Bank benthos consumption

The percent of total benthos consumed by the five predators considered for Georges Bank was notable, averaging ~13% across the time-series relative to the total estimate of Georges Bank benthos (70 g m^{-2}; Steele *et al.*, 2007; Figure 3.14). This is an artefact of the predators considered and use of a static value for total benthos availability; however, our results were similar to previous work of this region (Collie, 1987). It is of interest to include more benthivores and their consumption on Georges Bank, particularly species that have less or no commercial value, but are in high abundance or are major benthic predators (e.g. longhorn sculpin, little skate, and smooth dogfish). It is also of interest to consider applying these methods to additional regions of the Northeast US shelf.

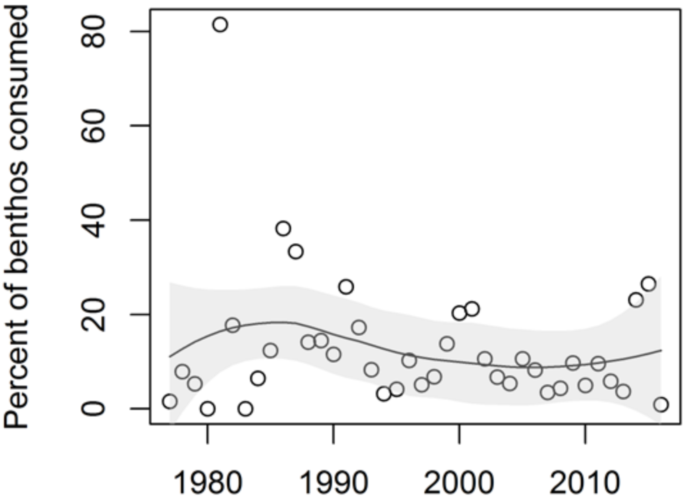

Figure 3.14. Percent of total benthos consumed on Georges Bank by winter flounder, witch flounder, American plaice, haddock, and cod. Smoother is LOESS with 95% ci, span =0.8.

Density-dependent growth

Density-dependent growth was apparent in both Georges Bank and Gulf of Maine haddock stocks, both of which have experienced dramatic recoveries since the late 1990s (only Georges Bank shown; Figures 3.15 and 3.16). Weak density-dependence was observed in Georges Bank winter flounder, but not in the other species stocks, most of which have declined in abundance and are currently at low density.

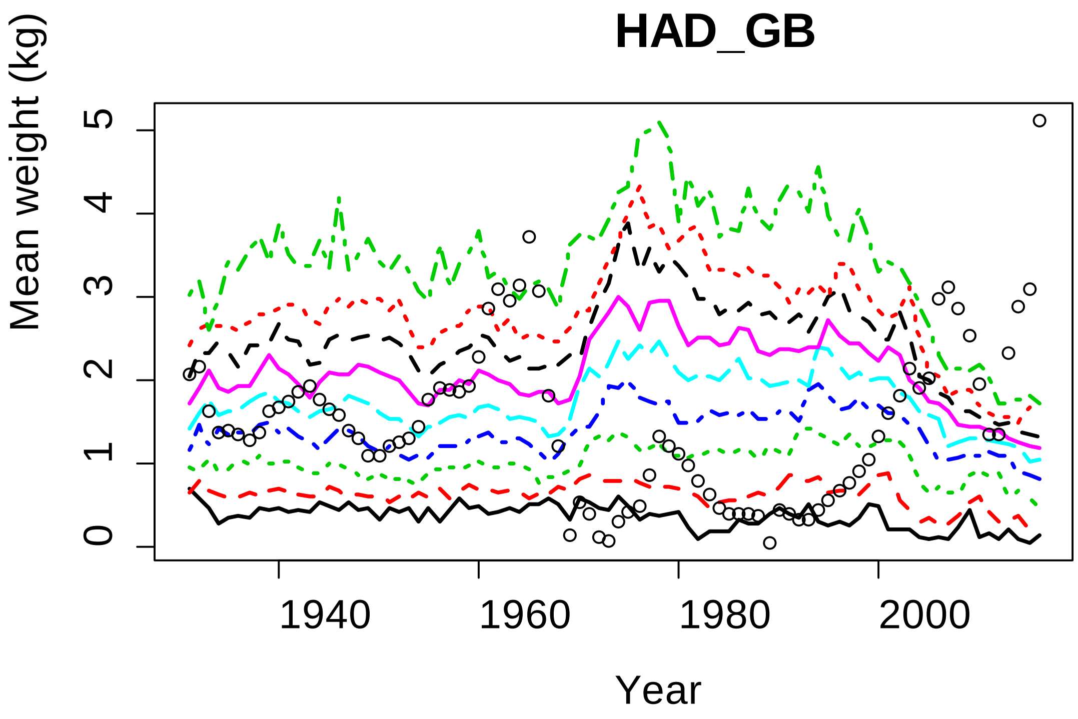

Figure 3.15. Mean weight-at-age of Haddock on Georges Bank. Each age is plotted as a line, ranging from age 1 (solid black line) to age 9 (broken green line). Circles indicate the spawning–stock biomass in each year.

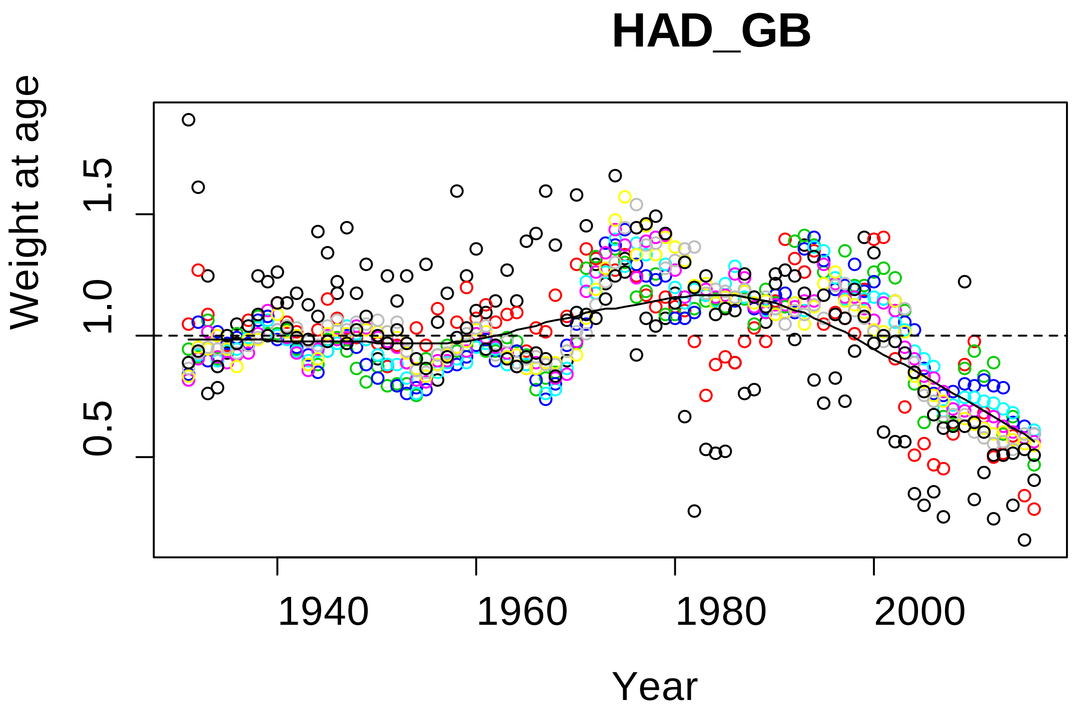

Figure 3.16. Standardized weight-at-age of Georges Bank haddock. Colour coding of each age is the same as in Figure 1. The solid line is a LOESS smoother (span=0.5) to indicate the temporal pattern.

Diet data were not available in all years, such that the model variants with Prey had fewer observations. Since Georges Bank haddock had the strongest effects of density and prey on growth, it is shown as an example.

Weight-at-age of Georges Bank haddock decreases with density (Figure 3.17) and increases with prey abundance (Figure 3.18). At high consumption (green line in Figure 3.17) there would be sufficient prey even at high density. At low consumption (black line in Figure 3.17) weight-at-age declines with increasing prey density. The highest density corresponded to low consumption, resulting in low weight-at-age.

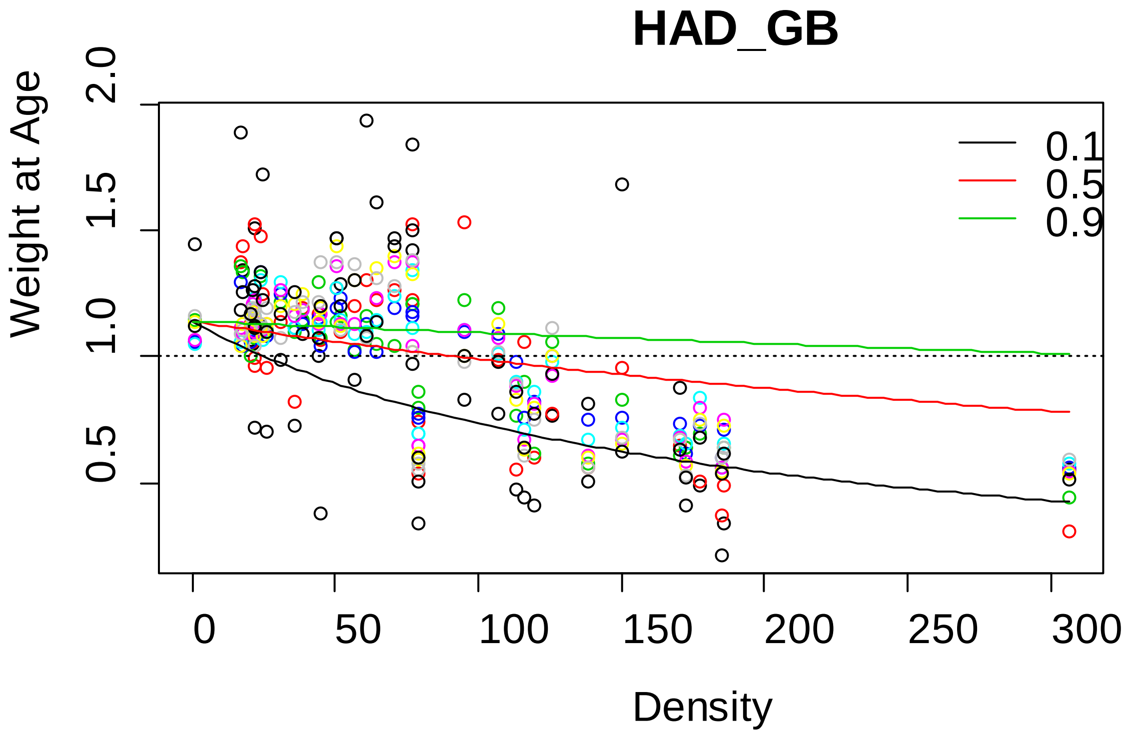

Figure 3.17. Relationship between weight-at-age and stock density (SSB) for Georges Bank haddock. Colour coding of each age is the same as in Figure 1. Solid lines indicate the effect of feeding level on weight-at-age. The different levels correspond to quantiles of stomach content per predator.

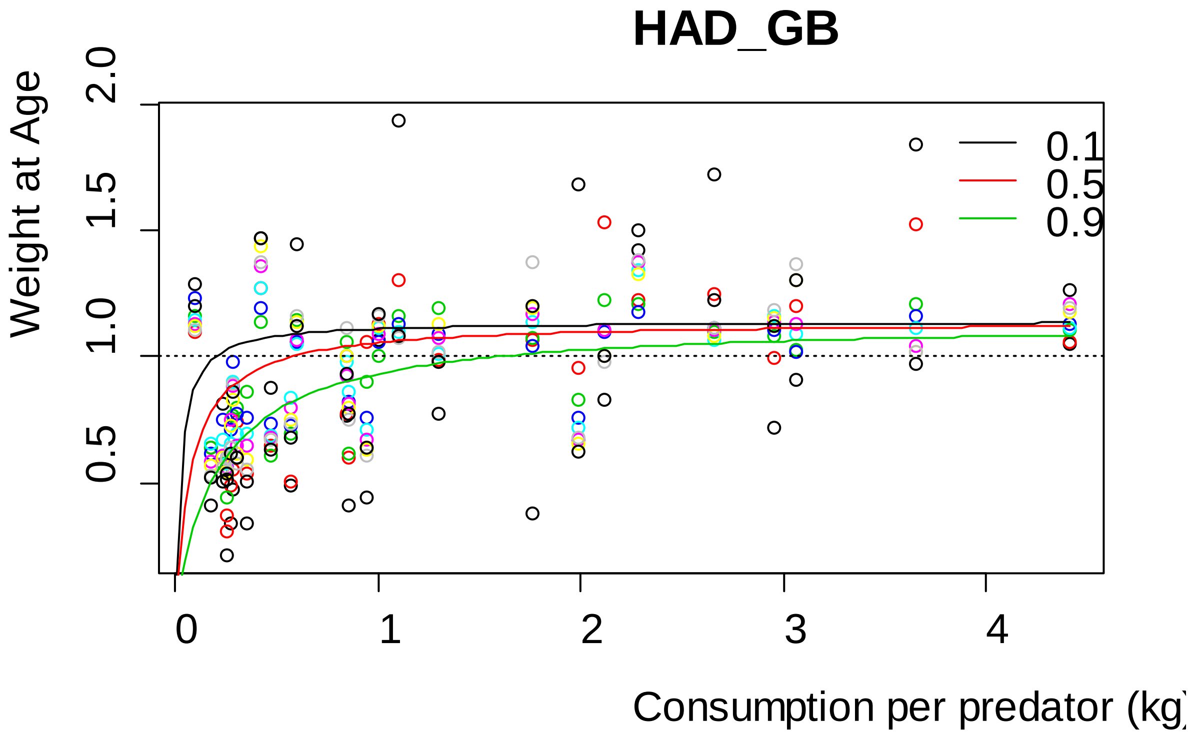

Figure 3.18. Relationship between weight-at-age and consumption of benthic prey for Georges Bank haddock. Colour coding of each age is the same as in Figure 1. Solid lines indicate the effect of stock density (SSB) on weight-at-age. The different levels correspond to quantiles of SSB.

Weight-at-age is an increasing function of consumption per predator (Figure 3.18). At low density (black line in Figure 3.18) this effect is only apparent at the lowest levels of consumption. By contrast, at high density (green line in Figure 3.18) this effect remains important for higher levels of consumption.

In summary, predator density and consumption per predator can independently affect growth rates. Density-dependent growth was apparent in the two haddock stocks that have recovered from low abundance to high abundance levels. For benthic-feeding fish this density-dependence is expected to result from food limitation. Benthic-feeding stocks that live in the same area and have a high degree of diet overlap share, to some extent, the same food resources. Stocks that live in the same area may exhibit food-dependent growth even if their abundance has declined. Therefore, we tested for the separate effects of density and food abundance and included stocks that had declined (e.g. flounder) together with those that have recovered (e.g. haddock). We found limited evidence of prey-dependent growth. This result may be due to the use of consumption as a proxy for prey abundance and the limited number of years with consumption estimates.

3.7 Recovery of baleen whales. Consumption rates and conflicts with fisheries

The recovery of baleen whales following the moratorium has been in many cases rapid, with some of the previously depleted species to be found in good numbers. As an example, data from the Trans North Atlantic Sightings Survey carried out within the

North Atlantic suggest that the abundance of fin whales increased from 15 200 to 20 600 between 1987 and 2007 (Víkingsson *et al*., 2015). The humpback whale is another example of a species that that has shown good recovery in many parts of the world. Within Icelandic waters, the humpback whale numbers were reported to increase annually by 12% over the period 1986–2001 (Pike *et al*., 2009) but levelled off with no clear trends afterwards. Bortolotto *et al*. (2012) reported a 26.7% increase in numbers of humpbacks whales in the SW Atlantic between 2008 and 2012.

The rapid recovery of baleen whale stocks has raised concern about their impacts of these on the structure of foodwebs and how their consumption of fish may affect the fishing industry. Studies that have examined feeding patterns of whales based directly on diet data are rare and are limited to nations in which whaling was allowed (e.g. Sigurjónsson and Vikingsson, 1997). In that study, they estimated the total consumption of minke, fin and humpback whale north of 60°N in Atlantic waters to be about 2.1, 2.6 and 0.2 million tonnes respectively based on the 1987, 1989 and 1995 sighting surveys and stomach data from harvested whales. The diet of fin whales consisted almost entirely of crustaceans, while the diet of minke whale consisted of crustaceans and fish in roughly equal proportions. There has been a decrease in numbers of minke whales within Icelandic waters, which could be linked to decreased occurrence of sandeel in the diets, accompanied with the reduced biomass of that fish species in the last decade (Víkingsson *et al*., 2015).

Overholtz and Link (2006) estimated the consumption on herring by marine mammals, seabirds and fish in the Gulf of Maine. The total consumption by these groups was variable but was estimated to be 310 000 t during the years 1998–2002. Of this total, demersal fish consumed the largest amount of herring, which peaked around 200 000 t during the period 1991–1994. The consumption of herring by fin and humpback whales rose over the study period (1977 to 2002). By 2002, the consumption of herring by the two species was estimated to be 41 000 t and 34 000 t, respectively. Moran *et al*. (2018) examined the predation by humpback whale on herring populations in three sites in the Gulf of Alaska. The consumption rates of was highly variable among sites, but was highest in the Prince William Sound, where it ranged between 21–77%. They concluded that the predation by humpback whales limited the recovery of depressed herring stocks at that site.

There has been some concern that whales are competing with the fishing industry for fishery resources. Surma and Pitcher (2015) used Ecopath models to examine the predation rates by several whale species. They concluded that both humpback and fin whales had top–down effects on the pacific herring biomass. They predicted that baleen whales could consume about 87% of the annual production of the Pacific herring (*Clupea pallasii*) after having attained full recovery. Smith *et al*. (2015 a) estimated the consumption of twelve marine mammal species off the NE United States. The diet of the baleen whales was dominated by shrimp (mostly euphausiids), (32% and 63% for minke fin whales respectively), zooplankton (42% and 85% for sei and right whales respectively), clupeids (30% and 27% for minke and fin whales respectively), and sandlance (16.5 and 15.5% respectively for fin and minke whales respectively). They estimated that the predation by marine mammals on the NE USA large marine ecosystem to be roughly equal to or higher than commercial catch for several commercially important fish groups, including clupeids, gadoids and flatfish. However, the authors stressed that for such comparison, the spatio-temporal overlap in the distribution of prey and predators needs to be considered. Ruzicka *et al*. (2013) concluded that whales and fisheries take similar proportions of the annual pelagic fish production (4–7%). They predicted that a fivefold increase in baleen whale abundance would reduce the production of competing planktivores (macro-zooplankton, euphausiids, carnivorous jellies, squid, forage fish) by less than 10%. Smith *et al*. (2015 b) examined how species interactions and potential changes to the ecosystem would change with various simulations involving catch recoveries of fish and rebuilding of the biomass of marine mammals. All models predicted an increase in marine mammal populations. Increase in the fishing mortality was predicted to slow down the population increase of marine mammals, because of reduced fish prey and greater incidence of accidental collision with fishing vessels. They further suggested that the predation by marine mammals (including baleen whales) could affect setting of reference points for those fish species that are commercially targeted.

3.8 Concluding remarks

Much of the work reported on this year is focused on documenting density-dependent growth reduction in response to recovery. We show that it occurs in plaice in the North Sea and the Eastern Channel, in North Sea dab and in haddock in the Gulf of Maine and winter flounder and haddock in Georges Bank. It is also shown for several other stocks in the ICES jurisdiction (Zimmerman *et al*., 2018). In Georges Bank Haddock, density-dependent growth did not occur in the 1960s, when the stock was at similar abundance as it currently is. We show that this may well have been a result of changes in benthos availability. We also show that consumption of benthos is highly variable for benthivores on the Northeast US continental shelf. Not only among species, but also over time. Interestingly, we found a generally decreasing trend in per capita consumption. Our estimate shows that the fraction of available benthos consumed by the studied Northeast US stocks is notable (~13%), but further work is needed to refine this estimate. We find limited evidence that density-dependent growth reduction in one stock carries over to other stocks with a high degree of diet overlap. On the basis of this year's work, we feel that it is safe to conclude that density-dependent growth reduction is a commonly observed pattern in stocks that show strong recovery. For the coming years, we will focus on understanding when it does (not) occur, and on its consequences for ecosystem and fish stock management. Finally, we have reviewed the potential effect of recovering whale populations on their prey, and find that this of such a magnitude that it may lead to conflicts with fisheries. We aim to explore this further.

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4 ToR b: Use empirical data and available multispecies models to examine how the degree of fisheries balance relates to ecosystem status

4.1 General comments

This ToR describes a multiannual workplan that was initiated at WGECO 2018. The first element *i* of the ToR was advanced in order to evaluate the impact of fishing on size structure of demersal fish communities. The effects of fishing on the foodweb were interpreted through an analysis of state and pressure indicators of Good Environmental Status (total biomass, total catch, size composition and species composition within demersal fish communities and within commercial catch). A methodology has been developed to set baselines for the species composition metric but further work is still required. Ideally baselines generated from a multispecies modelling approach would be contrasted to such empirical approaches to determine usefulness and robustness of any suggested assessment reference level.

4.2 Foodweb Indicators (state and pressure)

The following community level indicators were investigated by the group:

1) Total surveyed biomass of demersal fish (sum of survey swept-area corrected catch rates times by area surveyed);
2) Species composition within demersal fish communities (Mean maximum length, MML)

$$MML = \sum_j (L_{\max j} B_j) \Big/ B$$

where $L_{max,j}$ is the maximum observed length for species *j* and *B* is total biomass;

3) Size structure within demersal fish communities (Typical length, TyL)

$$TyL = \exp\left[\frac{\sum_{i=1}^{N} B_i \ln L_i}{\sum_{i=1}^{N} B_i}\right]$$

where *Bi* is the biomass and *Li* is the length of the *i*-th fish in a sample of *N* fish (WGECO 2014).

The second and third indicators above have been included in the OSPAR Intermediate Assessment and assessed based on trend only https://oap.ospar.org/en/ospar-assessments/intermediate-assessment-2017/biodiversity-status/fish-and-food-webs/. WGECO discussed and proposed procedures to develop baselines for these indicators given observed time-series. State indicators based on survey data were then contrasted to changes within the commercial catch (total catch and by size class). The commercial fisheries data were also used to evaluate the change in pressure indicators (i.e. Species composition within catch and Size composition within catch) by replacing the biomass *B* of species *i* in the above equations with the catch *C* of each species.

Foodweb assessment: proposed procedure to determine baselines for mean maximum length

Quality assured survey data were obtained from the Marine Scotland Science analysis of ICES DATRAS data http://data.marine.gov.scot/dataset/derivation-groundfish-survey-monitoring-and-assessment-data-product-northeast-atlantic-area. Analyses were made on North Sea IBTS data from Q1, Irish Sea GOV data (Northern Ireland) from Q1, Celtic Sea GOV data (Ireland) from Q4.

An overall assessment of a fish community requires multiple aspects to be investigated including: biomass of the community (total and by feeding guild), species composition and size structure within the community. Multiple indicators are required to complete such an assessment and each indicator should reflect change in a particular aspect of the community. The mean maximum length metric was trialled by OSPAR as an indicator to represent change in the species composition of fish communities. The indicator is high when the community is dominated by relatively large-bodied fish species (including elasmobranchs) that are particularly vulnerable to the effects of fishing. The pilot assessment of this indicator within the OSPAR intermediate assessment in 2017 was based on trends and relative changes with no baseline identified from which "good" status could be inferred. All regional seas within the ICES area are impacted by human activities and thus the species composition during pristine conditions cannot be observed. Multispecies models can provide estimates of the relative biomass of species within communities, under a range of fishing levels and environmental conditions, and a full investigation of a suite of models should be made to support the use of any empirically generated baselines and/or targets.

WGECO propose an empirical approach to identify baselines for the mean maximum length of demersal fish communities using data from trawl surveys of the North Sea (1983–2017), Irish Sea (1992–2016) and Celtic Sea (2003–2016). Given the long history of fishing in these areas and general depletion of large predatory fish (e.g. large sharks, skates and rays), it is reasonable to assume that each of these ecosystems can support large-bodied demersal fish species at their recently observed high (upper 25th percentile) biomass levels simultaneously. If this is not the case currently then this can be considered as a suitable aim for management (termed a "Recovery Goal" by WGECO 2017 for the species-specific abundance metric-level target within the sensitive species indicator). A system in which large species are heavily depleted cannot be considered to meet Good Environmental Status and further decline of vulnerable species should be avoided (i.e. species biomasses should be above the lower 25th percentile of observed biomass).

The distribution of biomass across species and the length at which species were considered "large," was investigated through the calculation of cumulative distribution functions of biomass against L_{max} (Figure 4.1). For the Irish Sea, the MML for the community based on species 25th and 75th percentiles of biomass is MML_{25} = 79 cm and MML_{75} = 80 cm respectively. For the North Sea, the two approaches both yielded the identical value of 77 cm. Given the strong similarity in MML_{75} and MML_{25}, for communities based on high and low biomasses, this shows that the species composition metric is not dependent on the overall biomass of the system. Here we propose to use the MML_{75} as representative of a system with a species composition in good state, but further investigation is required. In particular, the importance of single-species to the determination of the baseline should be further explored since this community-level indicator should ideally be driven by community level change rather than single-species effects. For instance, a large increase in the cumulative biomass function at L_{max} = 32 cm in the North Sea (Figure 4.1) is due to a large biomass of Norway pout *Trisopterus* *esmarkii* in the survey. How important is this species in fixing the MML from the percentile method?

A community where large species are depleted relative to the biomass of small species represents a system in poor status (i.e. where a high abundance of prey is available but the system does not support a high abundance of predators). Given our data, this can be represented by a cumulative distribution function where the species with L_{max} greater than the MML_{75} are depleted to their 25th percentile biomass levels while species with $L_{max} < MML_{75}$ are maintained at their 75th percentiles (Figure 4.2). This approach was trailed for the North Sea, Irish Sea and three distinct areas in the Celtic Sea (west of Ireland, west of Scotland and south of Ireland, Figure 3). The west of Scotland and south of Ireland areas demonstrate Good species composition, while the west of Ireland demonstrates a relative depletion of large demersal species. The conclusion for the west of Scotland area is corroborated by the increase in total biomass there (Figure 3). West and south of Ireland, the time-series of biomass are noisy but also show some evidence of an increase in biomass. The Irish Sea shows that recovery is underway in species composition but incomplete, while the North Sea appears to have stabilised near the upper reference level following a decline from a clear Good status in the early 1980s. The next section details our analyses of commercial data to understand the change in pressure on the system that may be linked to these changes.

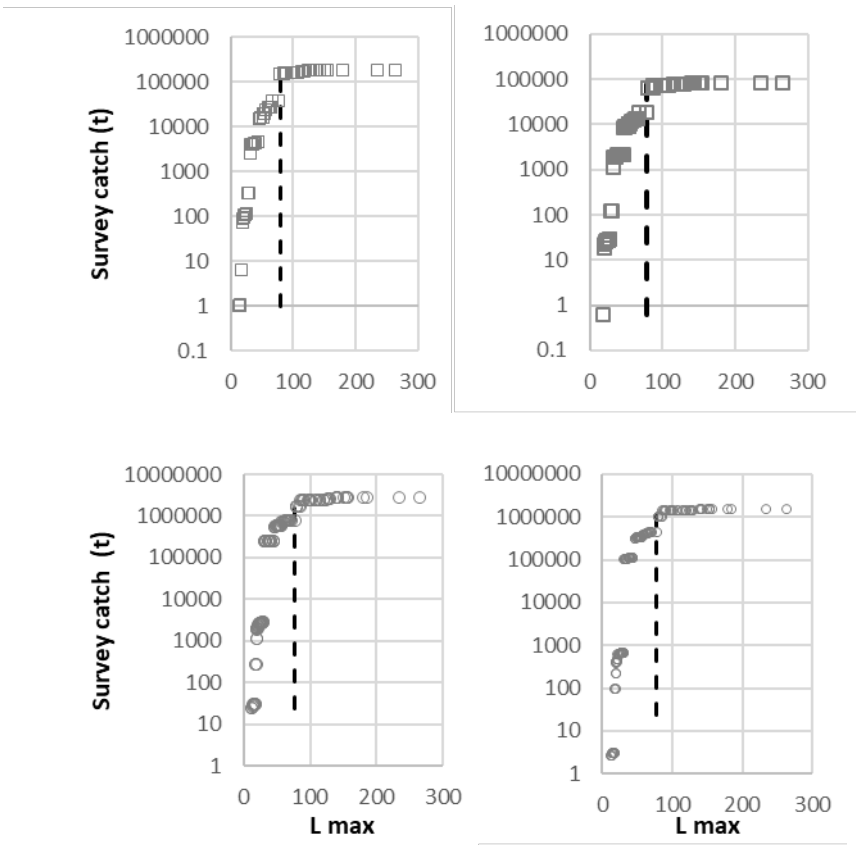

Figure 4.1. Cumulative distributions of log10 surveyed biomass (t) of demersal fish against L_{max} (cm). Top Irish Sea (squares) and bottom North Sea (circles), left based on 75th percentiles of biomass for each species and right, based on 25th percentiles. Vertical dashed lines show the mean maximum length of the community given the percentile data (i.e. MML_{75} and MML_{25}).

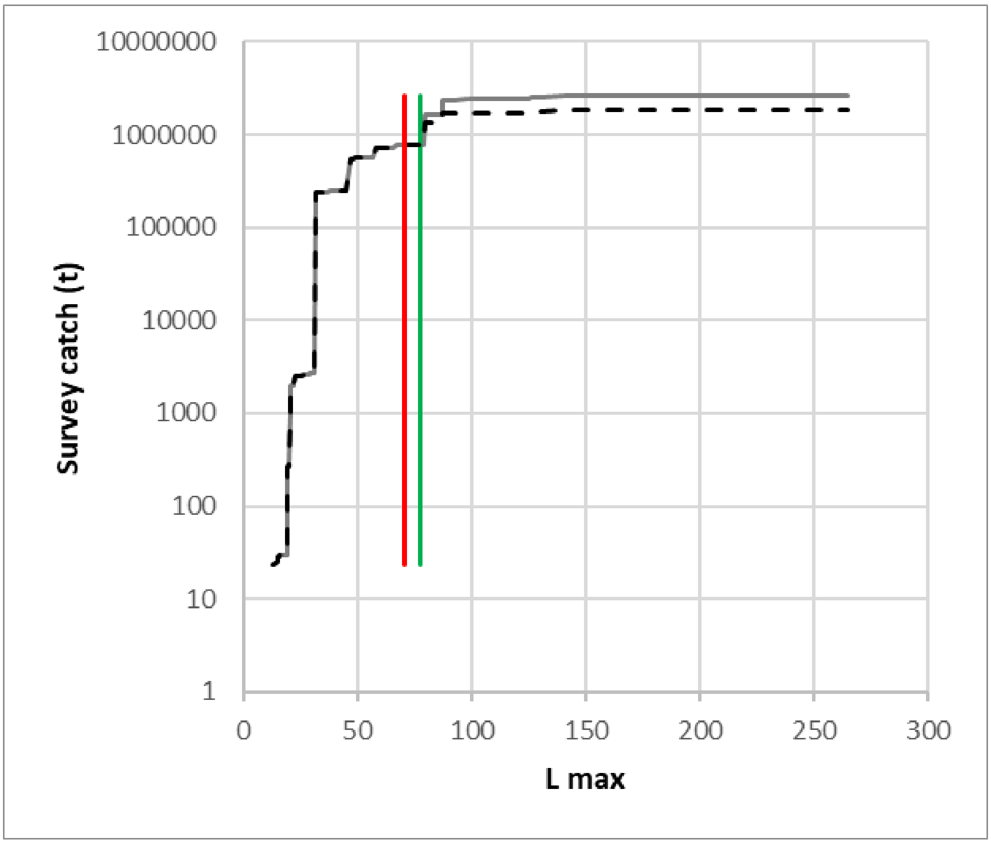

Figure 4.2. Cumulative distributions of surveyed biomass (t) of demersal fish against L_{max} (cm) in the North Sea. The green line is drawn at MML_{75} based on the cumulative distribution for all species at the 75th percentile and plateauing at the grey line (as shown in Figure 4.1 bottom-left). The cumulative distribution with depleted large species biomass is shown by the black dashed line and the MML of this curve is given by the red vertical line (71 cm).

Figure 4.3. Cumulative distributions (left) as in Figure 4.2 showing the depleted community (dashed lines, with corresponding MML red) relative to the high biomass community (solid lines with corresponding MML in green). Time-series of MML (right) shown with upper and lower baselines. Above the green line the community composition is considered in Good status and below the red area is a situation to avoid.

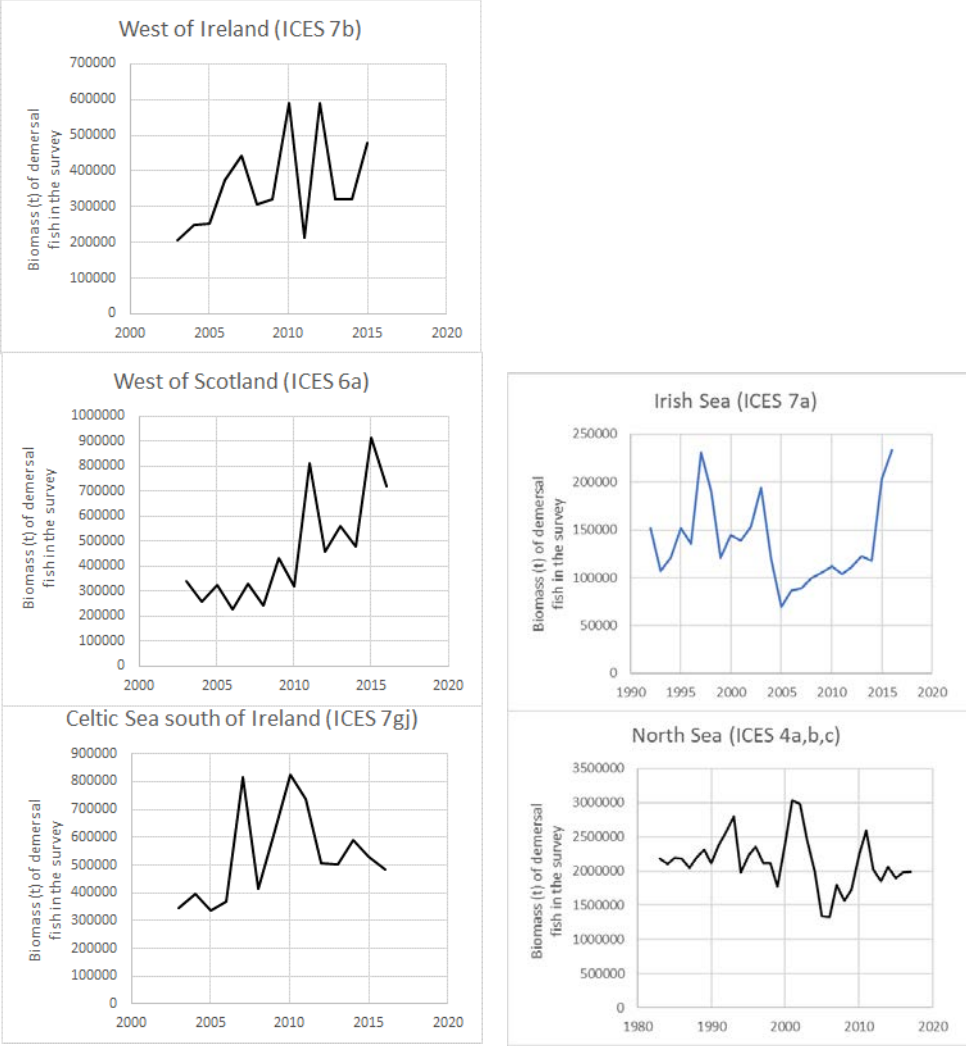

Figure 4.4. Survey estimated biomass of demersal fish per year.

4.3 Fisheries-dependent data and pressure indicators

Catch composition data from commercial fisheries were collated from the Observer Programmes for UK(E+W) and Ireland for the period 2007–2017. Preliminary investigations were made to determine the total annual catch, landings and discards by fleet and further studied to determine the catch within 20 cm size categories. For this purpose, species were classified into demersal fish, demersal elasmobranchs and pelagic fish. Pelagic elasmobranchs have only rarely been caught and thus were excluded from the analyses. The catch composition data considered included Irish fisheries using set gillnets (GNS), bottom otter trawls (OTB), fly shooting seines (SSC) and beam trawls (TBB) and UK fisheries using dredges (DRB), trammelnets (GTR), set longlines (LLS), OTB, midwater otter trawls (OTM), purse-seines (PS) and TBB. Data were analysed for five different areas comprising of the following ICES divisions: 6.a and 6.b ("West of Scotland"), 7.a (Irish Sea), 7.d (eastern English Channel), 7.e (western English Channel), and 7.b,c,f,g,j,k ("Celtic Sea"). Bottom otter trawls and beam trawls were the main gears in all areas accounting for 73–100% of the total catch within the period considered.

Preliminary visual data exploration of Irish and UK data suggested a reasonable consistency among datasets and thus they were analysed collectively. All gears were combined per area, and total catch as well as catch composition regarding the size distribution were investigated for the given time-series (Figure 4.5).

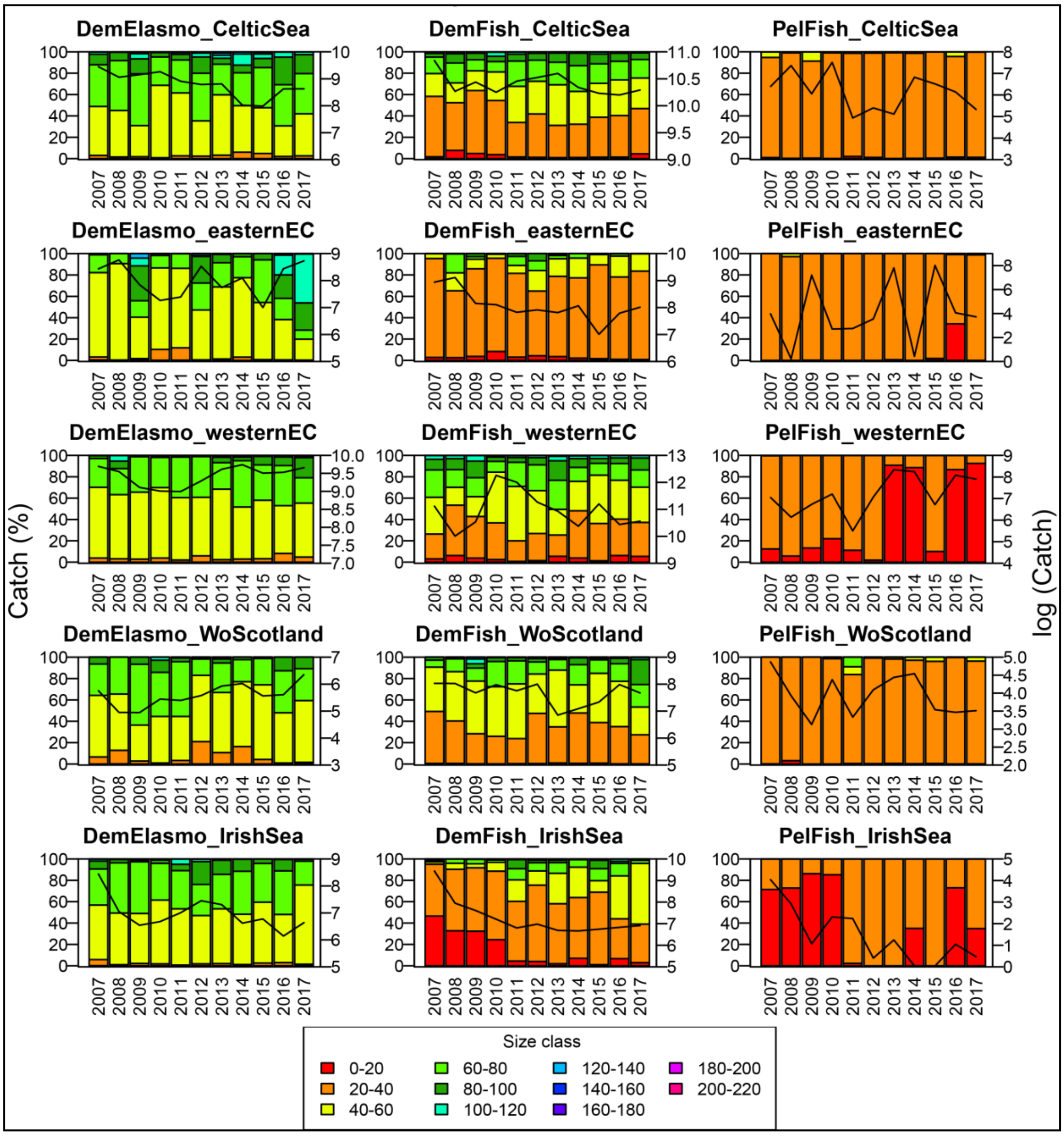

Figure 4.5. Catch from collated UK and Irish commercial fisheries measured as raised weight (t) for the Celtic Sea, the eastern and western English Channel, the Irish Sea and West of Scotland. Bars represent percentages per size class for each year and lines represent log-transformed total catch per year.

In the Celtic Sea, only slight changes in the size distribution of the three fish groups could be observed, though a general decrease in total catch of demersal elasmobranchs occurred over time until recent years. The survey demonstrates an increase in both total demersal fish biomass and MML following a reduction in the total catch in the area and in particular of large species. Significant negative correlations ($P<0.05$) between both commercial catch and MML in the survey the next year ($R = -0.72$, $n = 9$) (high catch leading to depletion of size structure) and MML of the commercial catch and biomass in the survey in the next year ($R = -0.69$, $n = 9$), (targeting large species leading to a decline in biomass) were found suggesting a strong pressure-state relationship.

In the eastern English Channel, however, demersal elasmobranchs showed a clear increase in the proportion of larger size classes since 2015, which coincides with an increase in total catch of demersal elasmobranchs since 2016. Considering the catch composition separately for the different gears used, the increase of larger size classes could be detected in beam trawls, set gillnets and trammelnets, which account for 62% of total catch in this area (Figure 4.6). Data on bottom otter trawls, however, showed little trend since 2015.

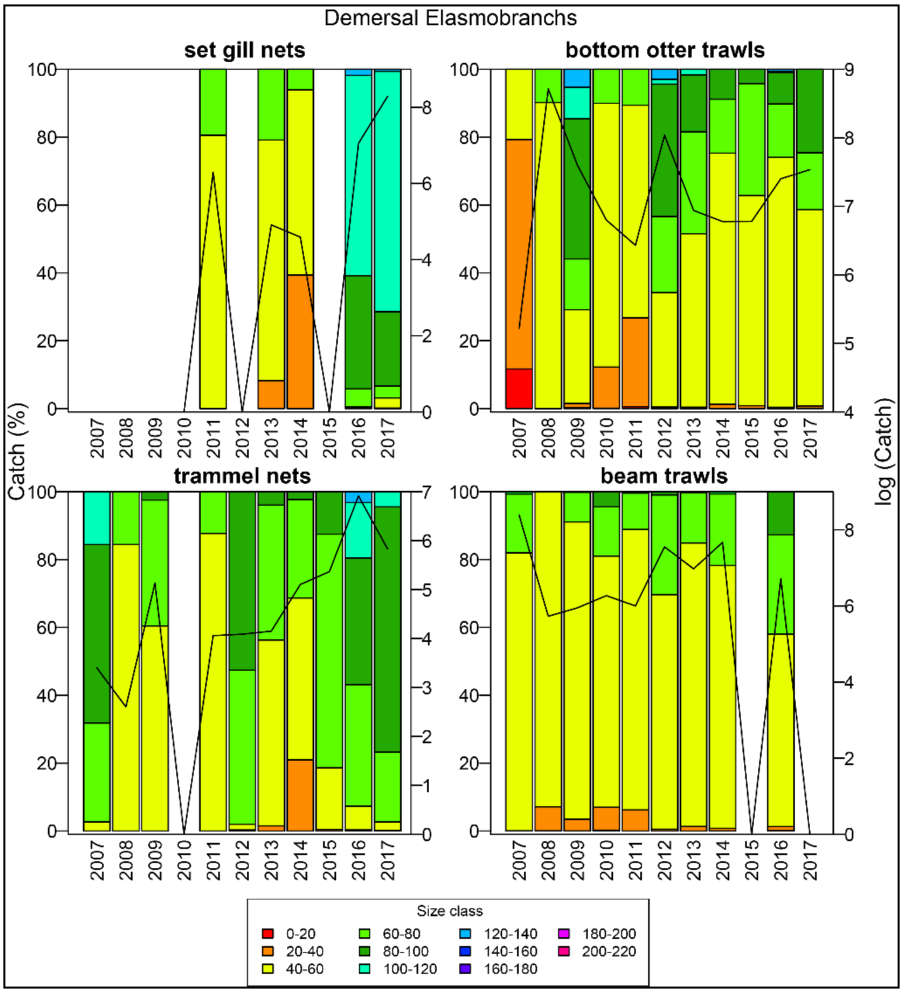

Figure 4.6. Catch of demersal elasmobranchs from UK commercial fisheries measured as raised weight (t) for the eastern English Channel per gear. Bars represent percentages per size class for each year and lines represent log-transformed total catch per year.

In the western English Channel, the proportion of larger demersal elasmobranchs in the catch increased only slightly over time. For demersal fish of the western English Channel, no clear pattern was apparent. For pelagic fish, high decreases of larger size classes were observed during the last years of the time-series apart from 2015. This can be explained by high total catches of sprat *Sprattus sprattus* in these years.

In the area West of Scotland, total catch of demersal elasmobranchs increased over time, while there was no clear trend in the proportions of size classes. For demersal fish, however, larger size classes increased in the catch composition in recent years.

Further investigations by gear could not be made as the bottom otter trawl was the only gear observed in this area. However, the larger proportion of large size classes appeared to be correlated with high catches in the ICES Division 6.a.

Finally, in the Irish Sea total catches decreased greatly over time for all fish groups considered, although a slight increase occurred for demersal fish since 2014. There was no apparent trend in size class composition regarding demersal elasmobranchs and pelagic fish, but larger size classes of demersal fish clearly increased throughout the time-series. The identified trend in demersal fish from the Irish Sea could mainly be attributed to the catch data from the bottom otter trawls, which closely resemble this pattern (Figure 4.7) and account for about 87% of total catch in this area. The change in size ranges present in the catch is due predominantly to a change in the species composition in the catch as indicated by the strong correlation between MML and TyL of the catch (Figure 4.8). Notably, the survey demonstrates an increase in both total demersal fish biomass and MML following the fall in the total catch in the area. However, correlations between either commercial catch and biomass in the survey in the next year ($R = -0.50$, $n = 9$) or MML in the catch and biomass the next year ($R = +0.51$, $n = 9$) are not significant ($P>0.05$).

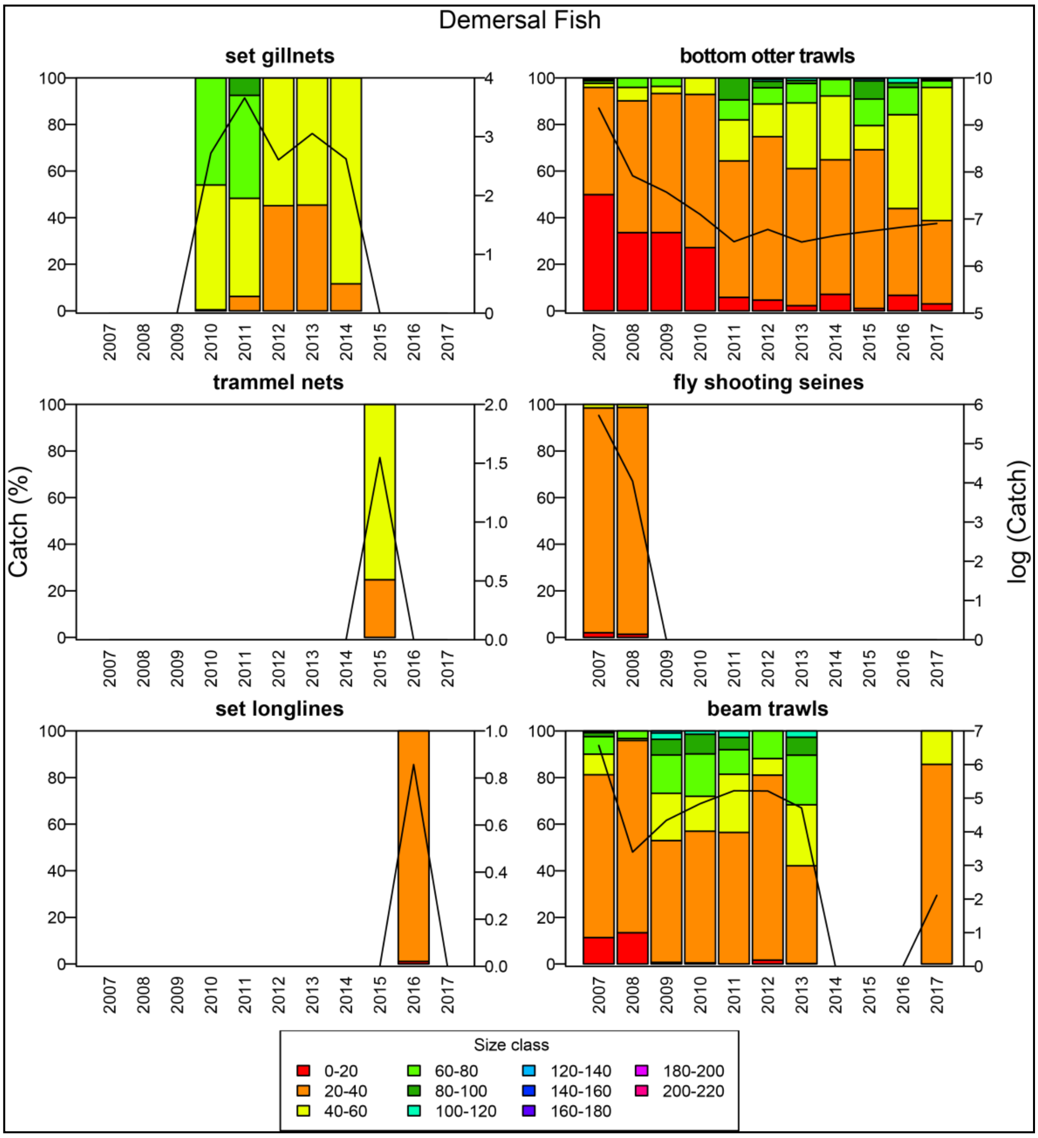

Figure 4.7. Catch of demersal fish from collated UK and Irish commercial fisheries measured as raised weight (t) for the Irish Seas per gear. Bars represent percentages per size class for each year and lines represent log-transformed total catch per year.

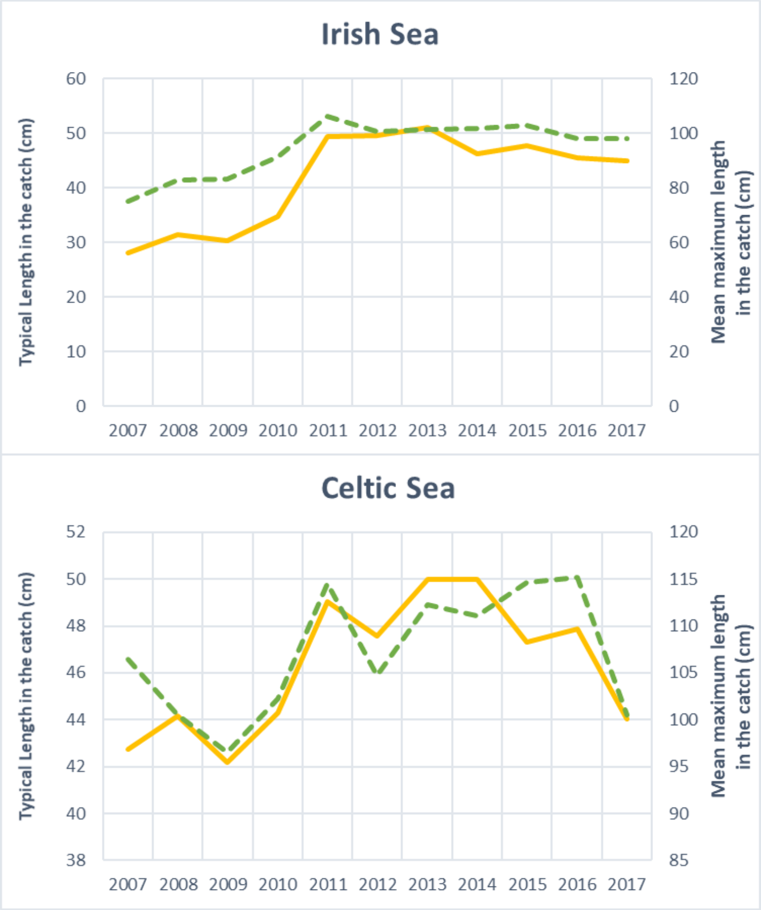

Figure 4.8. Mean maximum length (dashed) and typical length (solid lines) in the total catch from collated UK and Irish commercial fisheries data for the Irish Sea (top) and Celtic Sea (bottom).

4.4 Findings

A detailed, but preliminary, evaluation of observer data from English and Irish fisheries alongside the fisheries-independent survey data for the Celtic Sea and Irish Sea has shown that the species composition within demersal fish communities and the total biomass of the communities responds to the impacts of these fisheries (years with high commercial catch can lead to a depletion of size structure in the Celtic Sea, while, in the Irish Sea, a reduction in commercial catch was followed by a period in which the recovery of large-bodied species within the system was evident). Other nations' data should be combined with these commercial data to make a comprehensive assessment.

In summary, WGECO made progress in developing baselines for species composition assessment and this work will continue. Elements *ii* and *iii* will be progressed intersessionally with colleagues from WGSAM where possible.

5 ToR c: Examine individual species abundance trends to improve interpretation of assessment outcomes based on the "abundance of a suite of sensitive fish species indicator"

In order to facilitate the implementation of sensitive fish species indicators across the ICES area, WGECO discussed the general principles behind sensitive fish indicators and issues which require particular attention in order to ensure the quality and consistency of sensitive fish indicator evaluations.

The indicator "abundance of a suite of sensitive species" integrates three different components:

1) Identifying sensitive species;
2) Evaluating the state of sensitive species;
3) Integrating the evaluation of a suite of species into one indicator.

Each of these components was discussed in order to provide guidance for implementing and further developing sensitive fish indicators in future.

5.1 Identifying sensitive species

There are several methods available to define species sensitive to fishing (le Quesne and Jennings, 2012; Greenstreet *et al*., 2012) to ensure that only the most sensitive fish species are used and thus avoiding indicators that respond to a range of drivers and pressures other than fishing. Greenstreet *et al*. (2012) used estimates of L_{max}, L_{∞}, K, L_{mat} and the corresponding age at which 50% of the individuals reach maturity, A_{mat} and used a ranking procedure assuming that the most sensitive species were characterized by large values of L_{max}, L_{∞}, L_{mat}, A_{mat} and low values of K to rank species sensitivity. Le Quesne and Jennings (2012) used a fish life-history model and asymptotic length to predict sensitivity by estimating the fishing mortality required to reduce the stock to specific levels relative to the unfished status. Le Quesne and Jennings used a general relationship between size of the individual and the selectivity in the fishery experienced for that size. In addition to the results of these methods, a method based on Le Quesne and Jennings, but including catchability from Walker *et al*. (2017) was also available, referred to as the Gislason method here. The methods were compared to investigate whether they produced similar ranking of the species. It should be noted that the indices related inversely to sensitivity: sensitive species are expected to have low F_{lim} and low values in Gislason and in Le Quesne and Jennings, and a high sensitivity score in Greenstreet *et al*. (2012).

The r^2 between the estimates of the three methods were 0.41, 0.57 and 0.73, which reduced to 0.24, 0.55 and 0.56 after removing the highly sensitive Flapper skate (*Dipturus cf. intermedia*, Figure 5.1). In contrast, the four different estimates of sensitivity based on different levels of spawning stock depletion (e.g. F at maximum yield-per-recruit, F_{40}) given in Le Quesne and Jennings (2012) were all highly correlated ($r^2 > 0.98$ for all combinations). Presumably the differences between the results of Le Quesne and Jennings and of Gislason reveal the impact of including exploitation patterns specific to the species group. The Greenstreet *et al*. (2012) method is based on different principles and is approximately equally correlated to the other two methods.

The indices of sensitivity were not highly correlated to the corresponding sensitivity index F_{lim} of the assessed stocks (Figure 5.2). It should be noted, that the species available for comparison did only cover a part of the possible sensitivity ranges (i.e. the most sensitive and the least sensitive ones could not be included), thus explaining at least some part of the weak correlations. The main difference between the three methods is that assessment-based estimates use growth, maturity and natural mortality for relatively short periods and include stock–recruitment relationships, often also for a shorter time period. Particularly the inclusion of stock–recruitment plots have a large impact on F-based reference points. The species with the second lowest F(conS) is spurdog, where Le Quesne and Jennings predict a higher sensitivity than for cod whereas the other two methods predict a lower sensitivity than for cod.

Figure 5.1. Sensitivity estimated by Greenstreet *et al.* (2012) and Gislason (in prep) as a function of F(conS), the fishing mortality required to reduce spawning biomass per recruit to 10% of the value in absence of fishing (lower F(conS) equals higher sensitivity), from Le Quesne and Jennings (2012). Note that the indices related inversely to sensitivity: sensitive species are expected to have low F_{lim} and low values in Gislason and in Le Quesne and Jennings, and a high sensitivity score in Greenstreet *et al.*

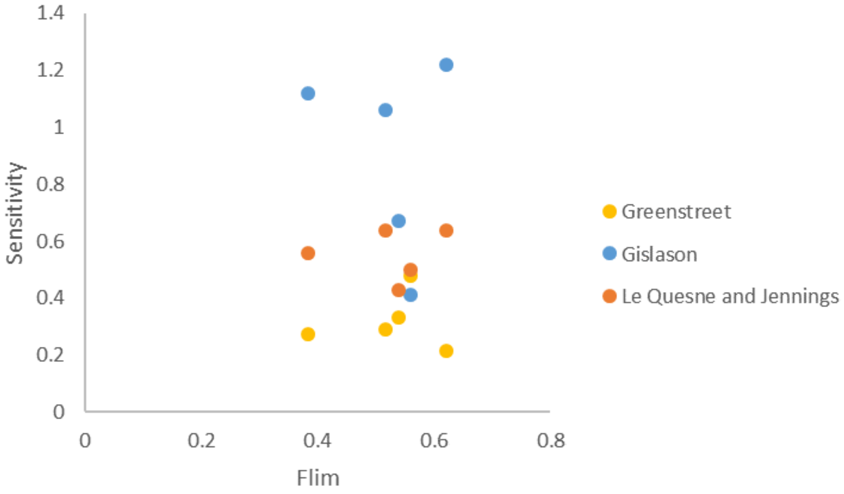

Figure 5.2. Sensitivity indices as a function of F_{lim}, the fishing mortality which on average brings the spawning stock to the limit reference point. Note that the indices related inversely to sensitivity: sensitive species are expected to have low F_{lim} and low values in Gislason and in Le Quesne and Jennings, and a high sensitivity score in Greenstreet *et al.*

Both methods aim to estimate the inherent sensitivity of species to fishing and consider species with life-history characteristics such as large asymptotic size to be more sensitive than species with smaller asymptotic size. While it is true that these species are inherently more sensitive, species with a lower inherent sensitivity can also be highly susceptible to fishing if they are exposed to a fishing mortality higher than the target species (e.g. the fishery for a widespread species concentrates in the primary habitat of a species with a more restricted spatial distribution). Such species may also need to be considered in the group of sensitive species in a region if they show evidence of being adversely affected by fishing in that area.

When defining sensitive species in an area, a key issue is which species should be included in the analysis of sensitivity. This is particularly important in transition areas e.g. between Lusitanian and boreal species or between low and high salinity tolerant species. In these areas, including species for which the abundance primarily reflects environmental conditions such as the inflow of warm or high saline water and for which the area is of minor importance in their life cycle, is likely to result in a list of sensitive species for which the effect of fishing may be masked by the effect of environmental changes. This can further result in a large number of false positive and false negative results. These considerations should obviously not be used as an excuse to exclude species where fishing is or has been the cause of their changed distribution (i.e. the area is not currently inhabited due to high fishing pressure). Hence, the list of species included in the estimation of species sensitivity should be carefully scrutinized to avoid e.g. stragglers and vagrant individuals or species as well as species at the border of their environmental tolerance.

Another issue discussed is how to define the limit between sensitive and non-sensitive species. Here, it may be worth considering that species which have been commercially exploited with a high level of catches for a long period of time are unlikely to be viewed as sensitive by non-experts and are furthermore generally managed through dedicated management measures and hence there is likely to be less of a need to monitor these species together with the sensitive species. Again, this principle should obviously not be used as an excuse to exclude species where fishing is or has been the cause of their changed distribution (i.e. the area is not currently inhabited due to high fishing pressure) or a sustained low abundance.

5.2 Evaluating the state of sensitive species

Sensitive species will generally be rare or low abundance species, at least in part due to exposure to historic high levels of fishing pressure albeit targeted on other species. That means that the data available are likely to be highly noisy, and more so the rarer the species is. Many surveys are not designed for any one species e.g. bottom-trawl surveys, and almost never designed for the rarer sensitive species. The gear may be poorly selective for that species, and the area covered may be inappropriate to the species distribution, migration or habitat. This can lead to highly variable abundance estimates, and this can be exacerbated by low population densities. For long-lived species, dramatic changes in abundance from year to year are unlikely to be compatible with their life history but can occur in surveys due to high annual variation in the availability and catchability of the species in the survey.

The indicators used for the evaluation of each species should be based on the best available knowledge for that species. This means that where approved and reviewed assessments exist, these should be chosen over time-series derived from, e.g. a single survey. If the assessments fail to include a particular time-series, it should be investigated whether this is due to lack of awareness of the time-series or lack of information in the time-series before deciding to include the time-series in a sensitive fish indicator. This means that a report of the state of sensitive species will need to be based on a combination of ICES assessment results and survey time-series, reflecting whether the species is assessed by ICES.

The time period and method to evaluate the indicator differs between implementations and areas. In the HELCOM area, the coastal fish indicators are assessed based on the presence or absence of significant trends in many areas, whereas in other areas, the level relative to a fixed indicator value is used. The method of Greenstreet *et al.* (2012) uses quantiles of the observed indicator values and hence continuously updates the reference level.

Due to the high variance in estimates of abundance for rarer species, using annual changes in e.g. catch rates is not likely to provide indicators robust to random sampling variation. The noisier the data, the longer the time period will be required to detect a change in the mean or a trend. This is likely to mean that rarer species require longer time periods to reduce the risk of numerous false positive and negative values (e.g. trends and means need to be estimated based on longer periods than five or one year, respectively).

5.3 Integrating the evaluation of a suite of species into one indicator

The indicator of a suite of sensitive fish integrates the indicator status for that suite of species, and compares that to a specific reference level for the integrated indicator. If other indicators exist that are later integrated with the indicator, an issue of weighting of the different indicators occurs. For example, if a general status is to be estimated for sensitive fish and coastal fish, the sensitive fish indicator may reflect 30 species whereas that of coastal fish may only reflect one species. An integration of these two indicators assuming equal weight to both may therefore not be appropriate. In general, the integration of indicators is a complex task requiring consideration of both statistical properties and management priorities. The topic is discussed in some detail in Section 6.2

in this report and will be addressed in further detail in WKDIVAGG and WKEXTINCT in 2018.

5.4 References

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6 ToR d: Investigate and report on potentially valuable ecosystem indicators for which full methodology has yet to be developed

In ToR d. WGECO was asked to: Investigate and report on potentially valuable ecosystem indicators for which full methodology has yet to be developed and propose methodologies and data sources. To include inter alia: Total mortality, Productivity of key predators, Primary production required to support fisheries, Guild level biomass, Total biomass of small fish, Pelagic-to-demersal ratio, and Benthic indicators. The current progress in the development of distributional indicators will be reviewed. Furthermore, this ToR should scope and evaluate methods to integrate indicators.

In this section we examine each of the specified indicators or types of indicator, review current progress on distributional indicators and examine approaches to aggregation of indicators.

6.1 Indicators considered

WGECO did not address benthic indicators in detail. Detailed evaluation was carried out in 2017 based on the report of WKBENTH (ICES, 2017a), and the review of that report by WGECO In the 2017 report (ICES, 2017b). Data and methods are fully available and have been described in detail, with worked examples in the BENTHIS project https://www.benthis.eu/en/benthis.htm

Some of the following text is partly based on text from WKFOOWI (ICES, 2014).

6.1.1 Total Mortality

This indicator, known as Total Mortality, Z (Fishing mortality + natural mortality) is commonly used in the ecosystem modelling community (e.g. Ecopath with Ecosim (EwE), see Figure 6.1.1) and is linked to productivity biomass ratios, PB. Its inverse, (1/Z) is an estimate of longevity.

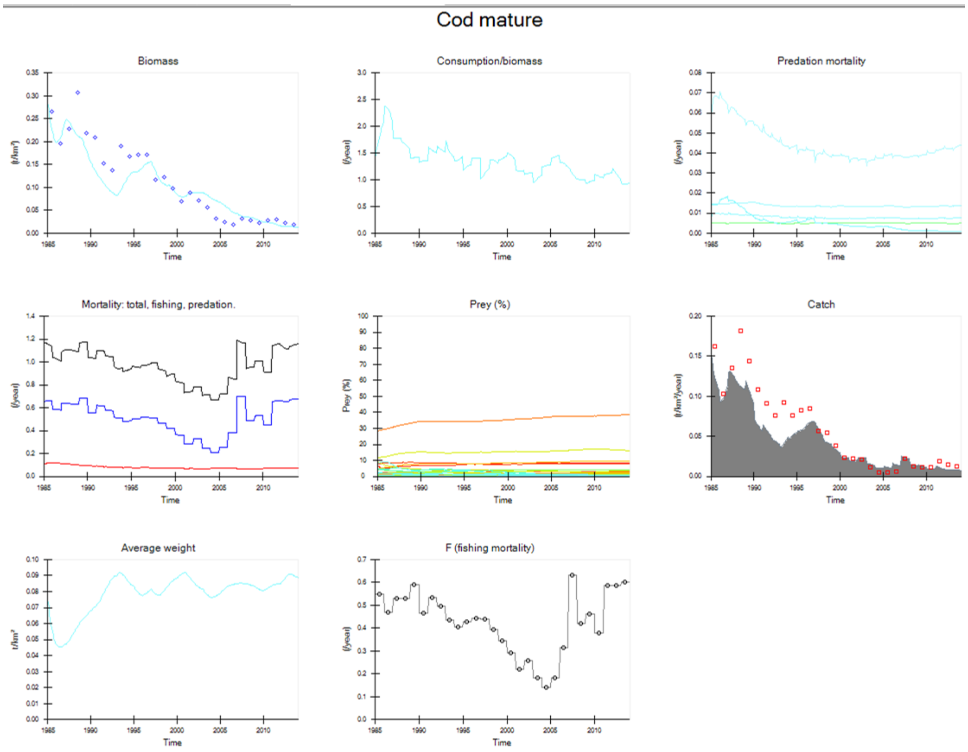

Figure 6.1.1. Typical single-species output from Ecopath with Ecosim, showing fishing, natural and total mortality for mature cod.

Total mortality determines year-to-year survival, affects resilience and long-term reference points such as F_{MSY} through the natural mortality incorporated. If mean weight of a species in the stock and catch, stock abundance and size structure remain constant over time, this indicator is conceptually equivalent to production/biomass PB ratio (Allen, 1971). Further, the inverse of total mortality is a direct indicator of longevity, which can be considered an indicator for resilience an indicator which is often more readily communicated outside the scientific community. It responds to management through the direct reliance on fishing mortality and the indirect effects of management of the abundance of predatory fish (WGSAM; ICES, 2012).

Total mortality is readily calculable from multispecies modelling, as most multispecies models explicitly model M and F. It would be expected to vary over years, subject to changes in F and M by species. In EwE the PB ratio is set for some species for the base year and modelled for others, but within some rule of thumb constraints (Heymans *et al.*, 2016). The indicator is sensitive to annual differences in diet composition and hence the accuracy of the indicator when estimated from model results relies on the ability of the model to predict diet composition in years without diet composition data. It should be noted that if estimated from EwE, the values are not directly comparable to those from stock assessments unless the model is defined with the same age stanza and the selection of age stanza included in the estimation are identical with those used when estimating average F in assessments. If using age-based multispecies models such as Gadget or SMS to estimate Z, the results retain the direct link to single-species assessment models and hence are directly comparable.

The annual anomaly in Z can be considered an indicator of the scale of pressure on the ecosystem, relative to the base year. When evaluated by WKFooWI it was scored high but was down-weighted due to difficulty in interpretation. WGECO (2017) thought it merited use, as methods are readily available for its calculation, and the problem of communicating the indicator to non-experts can be addressed by calculating the indicator as the inverse as (1/Z) as an estimate of longevity. Hence, WGECO considered this to be an indicator which has peer-reviewed data available in some areas, is potentially useful and can be estimated on a routine basis.

6.1.2 Primary Production required to support fisheries

Primary Production Required (PPR) is the primary production and detritus flows from trophic level (TL) 1 that are required to sustain fisheries (expressed as t/km²/year). This indicator allows the evaluation and comparison of fishing activities across ecosystems. The PPR is obtained by calculating the flows backwards, expressed in primary production and detritus equivalents, for all pathways from the caught species down to the primary producers and detritus. The PPR increases with biomass removed by fishing up to MSY if selectivity and species composition remain unchanged. PPR has been analysed also in reference to PP, to reflect a percentage of PP used to sustain catches.

Solar radiation is fixed by phytoplankton and provides energy for marine ecosystems. Subsequently, energy is transferred through foodwebs by predation and lost through metabolic processes. Ecosystem production results from the conversion of organic matter at each trophic level and depends on ecological features such as the number of feeding links, the efficiency of energy transfer from one trophic level to the next, and temperature (Chassot *et al.*, 2010). Production available to fisheries depends upon fishing mortality and targeted trophic levels in the foodweb. Fisheries focusing only on lower trophic levels may be energetically more efficient than those focused on top predators (Pauly and Christensen, 1995; Gascuel and Pauly, 2009).

$$\mathrm{PPR} = \sum_{i=1}^{n} \frac{C_i}{\mathrm{CR}} \times \left(\frac{1}{\mathrm{TE}}\right)^{(\mathrm{TL}_i - 1)}$$

PPR can be calculated from the equation above (Pauly and Christensen, 1995). Where C_i is the catch of species i, CR is the conversion rate of wet weight to carbon, TE is the transfer efficiency between trophic levels, TL_i is the trophic level of species i and n is the number of species caught in a given area.

Probably the key uncertainty is that this indicator requires estimates of transfer efficiency (TE) and trophic level (TL). TE is generally assumed to be 10–15% between trophic levels. Indicators of transfer efficiency themselves were not selected as indicators by WKFooWI for use immediately due to lack of data to systematically estimate TE. TL is equally challenging to estimate. It requires:

a) good quality, and regularly updated data on dietary relationships;

b) time-series of survey catch or landings from broad regional seas to avoid local population or fleet effects; and

c) accurate and agreed, regularly updated assessments of trophic levels.

Taken together, the uncertain validity of the TE estimates, and the complex data needs for TL calculation call into doubt the value of this indicator. As a broad, global indicator it may have value for comparing ecosystems, using generic values for TE and TL (Stock *et al.*, 2017). It may have less value at an individual ecosystem level, where temporal or spatial variability of TL for a given species may be substantial. It is possible to estimate a species TL using EwE for instance, and this might be a more tractable approach, but would also have to be used with caution.

There are other approaches to calculating this indicator. Maureaud *et al*. (2017) proposed an approach that did not use TL, but was based on the mean sea surface temperature, and K from the von Bertalanffy growth model. Jennings and Collingridge (2017) proposed a method using size spectrum, SST, and depth, although this was to link to primary consumers biomass, but could possibly be adapted for fisheries PPR.

WKFooWI considered that this indicator had a solid conceptual basis, and methods for its calculation exist, but the parameterisation is problematic. The difficulty of explaining the concept to the lay public is also a factor to consider. WGECO considered that the data are not yet available to support the assumptions of specific values of TE and therefore this indicator is not yet operational.

6.1.3 Guild level biomass

Biomass of trophic guilds is a measure of ecosystem structure, estimated as the aggregate biomass of each trophic guild. Individually they provide a measure of the change in biomass of trophic guilds. Collectively used they provide a measure of change in overall structure. It can be applied to all marine species if the information is available, based on survey data or model results. Work to date has largely focused on fish trophic guilds (Shackell *et al*., 2012; Rochet *et al*., 2013), but could be extended to invertebrates, birds, and marine mammals. Measures of functional diversity could also be developed using these data.

Methods for calculating these indicators are well developed and use survey and/or model derived data e.g. single-species assessments and/or EwE. WGECO (ICES, 2012) suggested that it was important to recognise ontogenetic changes in guild assignment, and a look-up table was established for most of the species in the IBTS. For instance, haddock (*Melanogrammus aeglefinus*) start (0–2 cm) as pelagic planktivores, from 3–59 cm as demersal benthivores, and from 60–112 cm as demersal piscivores. An example of the relative guild biomass from these analyses is presented in Figure 6.3.1. Changes in diets and hence guilds are further elaborated in Selden *et al*. (2018).

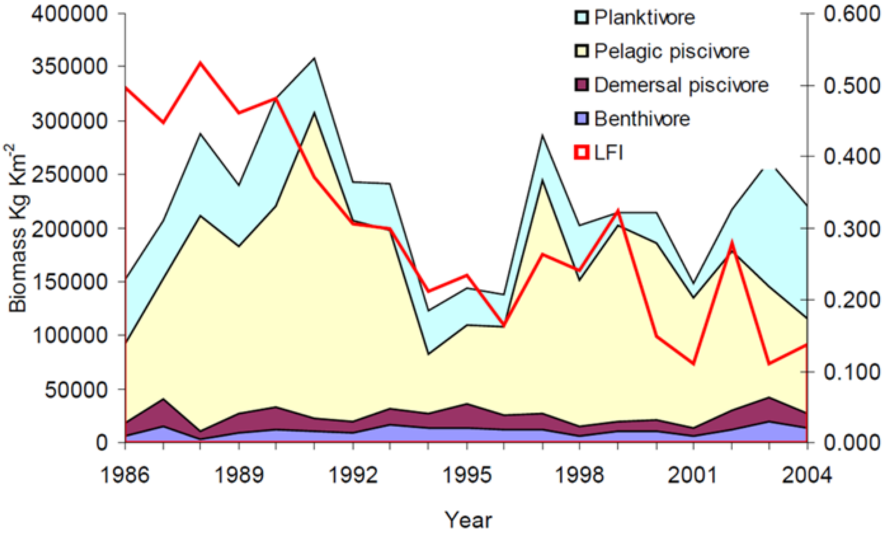

Figure 6.3.1. Trends in abundance (Kg Km^{-2}) of key functional groups of fish in the Celtic Sea fish community (WCGFS data).

The number of guilds used would also be a factor for consideration. Rochet *et al*. (2013) and WGECO used only four guilds, while Bulman *et al*. (2001) used ten guilds. Fewer guilds permits easier classification but may lump functionally dissimilar species.

Shephard *et al*. (2014a) suggested that this would best seen as a surveillance indicator rather than operational. It was considered as important to monitor the state of key functional components of the foodweb, but that there may be no strong scientific rationale for setting targets for trophic guild indicators. Other authors have advocated trophic guilds as units of fisheries management (Fogarty, 2014; Link, 2018).

WGECO considered this to be an indicator which has peer-reviewed data available, is potentially useful and can be estimated on a routine basis. Whether it is most appropriate to include only species that are e.g. planktivores throughout their life as done by Shephard *et al*. (2014b), or all species that are planktivores for part of their lives, as suggested in WGECO (ICES, 2012) should be examined further. Additionally, the issue of how species which are important in a guild but are not adequately sampled in trawl surveys should be investigated.

6.1.4 Total biomass of small fish

WKFooWI indicated that this indicator uses survey catch biomass of predefined small (pelagic) fish to assess exploitation levels of commercial stocks. The amount of energy transferred from zooplankton to higher trophic levels by pelagic fish is ultimately limited by the biomass of pelagic fish available. Shephard *et al*. (2014b) therefore suggest that both the biomass of individual stocks should be above precautionary reference points on average and the total-stock biomass of all pelagic fish together should be above a joint community reference point. In practice, the community reference point is always reached when all individual stocks are above precautionary reference levels. However, in the case where one or more stocks are substantially below single-stock reference points, additional care should be taken in the exploitation of the remaining stocks in the area.

This indicator uses survey and assessment data where available. It should be noted that small demersal fish may have a high sampling variability of the surveys, and also high interannual variability. The methodology accompanied with examples is presented in Shephard *et al.* (2014b). However, this approach was developed for pelagic fish, representing a possible guild biomass indicator, and linked to Pelagic/Demersal ratio indicator (below). As such, it is a viable and potentially useful indicator. It was also suggested that this could be extended to a total small fish indicator. This could be done by combining with, for instance, the small fish biomass calculated for the LFI, though this method will not address the issue of species with high abundance and low catchability (sandeel, sticklebacks, etc.). So in methodology and data terms, the indicator would be relatively easy to generate. What is not clear is what value the indicator would have, and how management could make use of it. "Small" demersal fish and small pelagic fish probably do not occupy similar ecosystem roles, although both could possibly be considered as "forage fish". Small pelagics are subject to targeted fisheries, while at least some part of the small demersal fish component are not. So management actions would be different, and choosing whether to target these on one or both of these components would be difficult.

In conclusion while the indicator is reasonably easy to calculate, it is difficult to see what value it would have, even as a surveillance indicator. As we already have, or could have, small pelagic and small demersal biomass indicators separately in the above section on guilds, combinations seems unnecessary.

6.1.5 Pelagic-to-demersal ratio

From WKFooWI - The ratio of pelagic to demersal fish (P:D ratio) obtained from fishery-dependent or -independent surveys is a commonly used metric that describes trophic energy flow and community structure (Caddy, 2000; de Leiva Moreno *et al.*, 2000; Rochet and Trenkel, 2003; Link, 2005). Changes in P:D ratio have been linked to anthropogenic pressures such as fishing and eutrophication. Targeted fishing can result in notable shifts in this indicator, however, changes may be not be entirely clear, as an increase in the P:D ratio could be caused either by an increase in pelagic fish or by a relative decrease in demersal fish. As an indicator of foodweb properties, P:D ratio may overlap with other large and/or forage fish indicators, but does capture important trophic relationships.

The indicator can be readily calculated from assessment and/or survey data where there are no analytical assessments, or from guild biomass indicators. There may be issues where either method calculates SSB rather than Total-stock biomass, but it should still represent the relative balance between the two components. Hence WGECO considered this to be an indicator which has reviewed data available, is potentially useful and can be estimated on a routine basis.

6.1.6 Productivity of key predators

These indicators have been proposed as part of the MSFD since inception (Rogers *et al.*, 2010). They can include, mammals, seabirds and fish (Wanless *et al.*, 2005; Soto *et al.*, 2006; Litzow *et al.*, 2006). They are often represented as some form of breeding success indicator, available from surveys, and relative to adult numbers, or for predatory fish, condition or weight-at-age. They can also be considered as indicators of the availability of food, with the inference that a healthy top predator population requires and represents a "healthy" foodweb. In a negative sense, this is likely to be correct, if there is a shortage of prey, this will likely negatively impact top predator breeding success. The converse is less likely to always be the case. Predator breeding failure or growth changes could be as a result of lack of prey, but could have numerous other causes.

Data and methods exist to develop these indicators, and they are already implemented in legislation. WGECO considered this to be an indicator which has reviewed data available, is potentially useful and can be estimated on a routine basis.

6.1.7 Distributional Indicators

WGECO looked at distributional indicators in the 2016 report (ICES, 2016-Section 6). Indicators were grouped into five topics: geographical, occupied area, aggregation, pattern and pattern dynamics. Detailed indicators within each of these categories were described along with data sources for their calculation. It was noted that distributional indicators were particularly vulnerable to survey coverage, and changes therein. It was also noted that these indicators can show changes as a result of more than one pressure, so could lack specificity. By the same token, they may be difficult to interpret for management action. Where the indicators were for geographical regions or ecoregions, it was recognised that they would be difficult to interpret where there were animals whose distributional range was greater than the region under consideration. They were broadly seen as most useful as surveillance indicators.

The Working Group on Fish Distribution Shifts (WKFISHDISH-ICES, 2017) used three main types of approach.

- Presence-absence; based on survey data and by ICES rectangle;
- Biomass trends; changes in biomass between two management areas over time were explored by investigating trends in survey abundance over time;
- Calculating the centre of gravity of a stock within a survey region; using a spatial model fitted to each year and survey.

These approaches were also included in the WGECO appraisal. A review of the literature did not uncover any other new approaches to this subject.

WGECO considered these indicators to have reviewed data available in some cases, potentially useful and that they can be estimated on a routine basis.

6.2 Methods to integrate indicators

Two approaches to integrating signals from indicators can be imagined. One would be using integrative indicators, the other would be to integrate the signals from multiple indicators. Many of the indicators described above are integrative. For instance, a demersal-to- pelagic ratio integrates the biomass from two different compartments. In turn, these biomass values would integrate the biomass of each individual species, or possibly guilds, in the two compartments. The guilds also integrate the species biomass. These are useful, derivatives from the initial basic biomass indicators and can convey new information, but are all based on broadly the same information collected on the same basis. Integrating indicators that do not convey the same type of information is a more complex problem, and cannot be accomplished simply by adding them together. For example, combining fish stock biomass with seal pup production or plankton lifeform indicators would be impossible. What can be combined are the signals provided by these indicators. If the indicators have a reference value, it can be over or undershot. For instance, a biomass reference value that was undershot would be an alarm signal, when overshot it would not be.

Combining information from multiple, and possibly conflicting, ecological, and additional relevant indicators is a general problem for sustainable resource management (Campbell *et al.*, 2002), and is especially relevant to fisheries (Alder *et al.*, 2010) as well as other sectors such as forestry (Wolfslehner and Vacik, 2011). Piet and Rice (2004) and Duggan *et al*. (2015) recognized this as a threshold response mode of fisheries management, amenable to the Signal Detection Theory (SDT; Egan, 1975), used to quantify the probability that an observer (operator) may respond when thresholds are exceeded. Duggan *et al*. (2015) took an analogous approach, which was similar to receiver–operator characteristics (ROC; Metz, 1978; Søreide, 2009) to quantify the evidential support behind management decisions. The method has also been applied recently to zooplankton indicators (Jernberg *et al*., 2017).

Another approach was taken to combine indicators for a range of sensitive species (Greenstreet *et al*., 2012; ICES, 2017). In this approach, the aim was to decide how many sensitive fish species abundance indicators might be expected to be declining at any one time simply due to random variation based on demonstrating significant departure from the binomial distribution. Another approach would be Integrated Trend Analysis (ITA) used for Integrated Ecosystem Assessment (IEA). This would use many indicator trends and use PCA (Kenny *et al*., 2009) to find the main multi-indicator components. A more advanced approach is described in Planque and Arneberg. (2018). Some IEA methodologies (Harvey *et al*., 2017) also integrate indicators and other information. For instance, the ODEMM methodology (Knights *et al*., 2015) integrates empirical and modelled data and indicator series as well as expert judgments. Finally, an evaluation of methods for aggregation was also included in the WGBIODIV 2015 report (ICES, 2015). This mainly focused on One-out-all-out, and a range of averaging techniques, the implications of each approach was described in some detail.

The key element of integration of indicators etc. is that it be useful in a management and policy-making context. WGECO suggested that this could be a full ToR for a future meeting, but this would require some representation from the likely users. This role could be played by secretariat personnel in the first instance.

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7 ToR e: Prepare a list of topics to be considered by WGELECTRA

7.1 General remarks

The request of the Dutch ministry focuses on ecological and environmental impacts of pulse trawling and beam trawling when exploiting the TAC of the North Sea stock of Dover sole (*Solea solea*). There are three pulse systems in use or at least being developed for North Sea fisheries. They differ by target species: flatfish (Dover sole) vs. Brown shrimp (*Crangon crangon*) vs. *Ensis* spp. Shrimp, *Ensis* and flatfish-directed pulse trawls have different pulse frequencies and hence may have differential effects of electricity to consider. The request is directed towards a comparison of pulse trawls targeting Dover sole in the North Sea to traditional beam trawls. Sole can also be a bycatch species in the shrimp-directed pulse trawl fishery. WGECO does thus not consider potential impacts on regions outside the North Sea or from the pulse fishery targeting *Ensis*. The questions are organized under the five topics given by the ToRs.

7.2 The sustainable exploitation of the target and bycatch species (species and size selectivity)

1) Species and size selectivity were considered by Van Marlen *et al*. (2014), whose study was based on three fishing trips. What is the actual effect of electricity on the catch of the target species (sole) and bycatch species like European plaice (*Pleuronectes platessa*)? Van Marlen's study, and others to date, have been mostly catch comparisons (i.e. one gear compared against another), which do not measure the intrinsic selectivity of the net. There is a need to complement these with a formal selectivity study for a standard pulse-trawl configuration (Wieman *et al*., 1996).

2) The spatial distribution of the fishing effort of the Dutch tickler-chain trawls (Figure 7.1, left panel) has changed when pulse trawling was introduced (Figure 7.1, right panel). What are the consequences of this shift? Possible research questions are:

  2.1) How has the cpue of sole (and bycatch species) changed in space and what are possible consequences of this shift in species and size catch composition on populations?

  2.2) Pulse trawls can be used in softer sediments compared to beam trawling. Does this have consequences on refugia for sole?

  2.3) How has cpue of sole (and bycatch of other fish species) changed over time since the introduction of the pulse trawl? Has catch efficiency increased due to pulse trawling, i.e. are pulse trawls able to catch more and/or larger sole as opposed to conventional beam trawls? Do pulse trawls maintain catch rates of larger sole since their introduction? A time-series of cpue by size class and species could indicate how cpue changes since the introduction of pulse trawling.

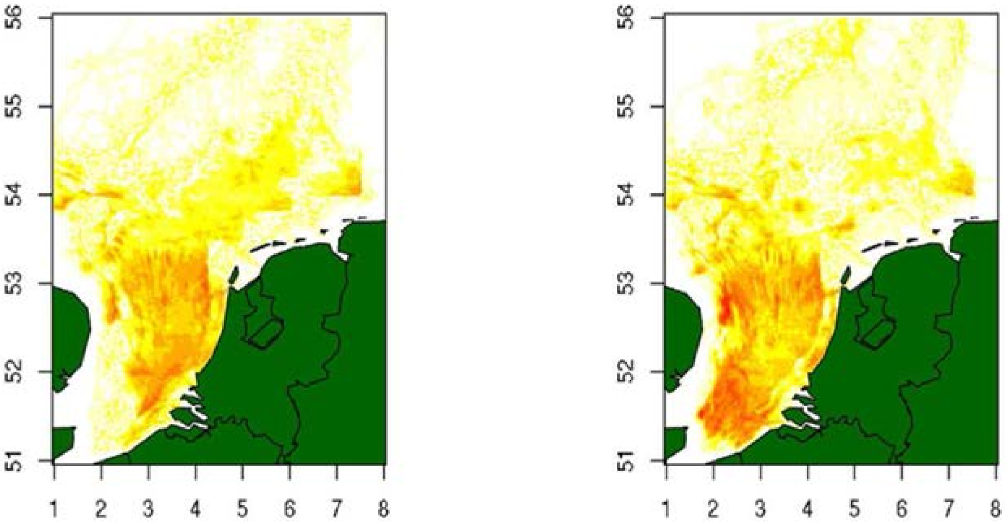

Figure 7.1. The spatial distribution of the fishing effort with Dutch tickler-chain trawls (left panel) and the pulse trawling (right panel).

7.3 Target and non-target species that are exposed to the gear but are not retained (injuries and mortality)

1) Laboratory experiments have been conducted to test for the effects of electricity on several species and life stages. How comprehensive and representative are these experiments with respect to evaluating the impacts on different life stages of marine organisms? Should the effects on more species or size classes or life-history stages be examined? Are there critical life stages that are at risk, such as metamorphosis of flatfish or the period of gametogenesis?
2) The effects of electric pulses were studied for Atlantic cod (*Gadus morhua*), showing significant effects (De Haan *et al.*, 2016; Soetaert *et al.*, 2016). What are the potential risks of these injuries to cod populations?
3) Can the results from laboratory experiments be extrapolated to field settings? Are there delayed effects (e.g. on growth, reproduction, etc.)?
4) Investigate in more detail the impacts of effects of pulse fishing on electrosensitive species. While no effects of pulse trawling have been shown for lesser spotted dogfish *Scyliorhinus canicula*, the effects of pulse fishing on electrosensitive species (e.g. sharks, skates and rays) in the sole directed fishery are still poorly understood.

7.4 The mechanical disturbance of the seabed

1) Local reduction of mechanical disturbance by replacing tickler-chain trawls with pulse trawls is expected (Depestele *et al.*, 2016). Were the sediment, habitat and hydrographic conditions investigated representative of the main areas where the fleet operates? Which aspects were (not) covered? What can be said on seabed impact at the fleet level (e.g. taking into consideration the change in gear use and effort as well as the displacement of effort)?
2) What are the effects of pulse trawling on the geochemistry (e.g. redox potential) and sediment properties in areas where fishing can be carried out by these pulse trawls but not beam trawls.

7.5 The structure and functioning of the benthic ecosystem

1) It is important to obtain comprehensive information on the effects of pulse trawling on benthic communities in the field. Can we extrapolate the effects of electricity derived from laboratory studies to field settings?
2) There is a need for a trawling experiment that will compare the effects of both pulse trawl and beam trawl on benthic invertebrates simultaneously, using rigorous replicated design (e.g. BACI) and including estimates of the benthos bycatch from these experimental fisheries. Conduct a power analysis to estimate the effect size that could be detected.
3) What is the fate of the non-target species in the path of the trawl but not retained? Direct sampling in the tracks of beam and pulse trawls can be conducted with divers or with directed grab sampling.
4) Evaluate the behavioural responses of infauna to electrical stimulation. Is infauna stimulated with electricity more prone to come to the surface where it is more likely to be predated upon?

7.6 The impact of repetitive exposure to the two gear types on marine organisms

There is some concern over the longer term impacts of repetitive sublethal exposure of benthic organisms to electric pulses. Cumulative effects of electric stimulation over longer time periods differ from the cumulative effects of physical disturbance that result in mortality. The evaluation of cumulative risk could involve dose–response experiments in the laboratory (e.g. growth rates, stress) and spatial distribution modelling to estimate the probability of encounter. Given the patchiness of trawling in time and space, there will be a need to define repetitive exposure with a time-scale relevant to the stress variables studied to establish the magnitude of any effect.

7.7 References

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Wieman, D.A., Ferro, R.S.T., Fonteyne, R., and Millar, R.E. 1996. Manual of methods of measuring the selectivity of towed fishing gears. ICES Cooperative Research Report 215.

Annex 1: List of Participants

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Annex 2: Meeting Agenda

	Thursday April 12th
09:30	Opening of the meeting
	Adoption of ToRs and Agenda
	Assignment of ToR leaders and subgroups formed Overview of presentations
11:00	Coffee
11:30	Initial discussion of ToRs a and b
	Initial discussions on ToR d
13:45	Lunch
15:00	Initial discussions on ToR e
17:00	Coffee
17:30	Preliminary workplan presented
18:00+	Adjourn
	Friday April 13th
09:00	Start
09:30	Subgroup work
1100	Coffee
11:30	Subgroup on A
13:00	Skype on ToR E
13:45	Lunch
1600	Subgroup on B
17:00	Coffee
18:00+	Adjourn
	Saturday April 14th
09:00	Plenary: status update from all subgroups
11:00	Coffee
	Subgroup work
13:45	Lunch
15:00	Plenary: initial discussion of ToRs and scheduling of 2019 WGECO meeting
	Subgroup work
17:00	Coffee
18:00+	Adjourn
	Sunday April 15th
09:00	Subgroup work
11:00	Coffee
	Subgroup work
13:45	Lunch
15:00	Field trip
	Monday April 16th
09:00	Plenary: Finalize the list of topics to be considered by WGELECTRA (ToR e)
11:00	Coffee
	Drafting session/subgroup work
13:45	Lunch

15:00	Plenary: report and discussion of subgroup work
	Drafting session/subgroup work
17:00	Coffee
	Drafting session/subgroup work
18:00+	Adjourn
	Tuesday April 17th
09:00	Subgroup reporting ToRs a–e
11:00	Coffee
	Drafting session/plenary/subgroup work
13:45	Lunch
15:00	Reconvene
	Drafting session/plenary/subgroup work
17:00	Coffee
18:00+	Adjourn
	Wednesday April 18th
09:00	Plenary to review draft report sections
10:30	Coffee
	Drafting session/subgroup work
13:45	Lunch
14:00	Plenary: ToRs and scheduling of 2019 WGECO meeting
15:30	Coffee
	Drafting session/subgroup work
18:00+	Adjourn
	Thursday April 19th
09:00	Plenary to review Final Report sections
10:30	Coffee
	Tidying up loose ends
13:00+	Adjourn

Annex 3: WGECO terms of reference for the next meeting

The **Working Group on the Ecosystem Effects of Fishing Activities** (WGECO), chaired by Jeremy Collie, US and Stefan Ragnarsson, Iceland, will meet in Copenhagen 9–16 April 2019 to:

- a) Investigate the ecological consequences of stock rebuilding, with particular emphasis on benthivorous fish and invertebrates.
  - i) Make first-order estimates of predation pressure on benthos;
  - ii) Examine evidence of food limitation and density-dependent growth;
  - iii) Compare the footprints of trawling to the footprints of predation pressure on benthos.
- b) Use empirical data and available multispecies models to examine how the degree of fisheries balance relates to ecosystem status.
  - i) Compare the length composition of total catch (landings and discards) to the length composition in the survey for one region (e.g. Irish Sea);
  - ii) Use multispecies models (developed by WGSAM) to identify targets for ecological indicators of state (i.e. status) that relate to an acceptable risk of species diversity loss; and
  - iii) Use output of multispecies models to investigate how proposed management strategies affect fisheries balance.
- c) Review the knowledge of spatial distribution indicators for fish and benthos.
  - i) Make recommendations on which indicators to develop, considering both how useful/important these are, and also simplicity of use and clarity of communication;
  - ii) Test several candidate spatial distribution indicators; and
  - iii) Scope and evaluate methods to integrate indicators.
- d) Conduct a “reality check” and horizon scanning survey within WGECO. The aim is to develop a consensus view of the major emerging issues in relation to fisheries and ecosystems, and on which WGECO could focus future work. WGECO members will provide a list of emerging issues (horizon scanning), that would benefit from scrutiny by WGECO. This list will be collated and used as material for a plenary discussion, and with the aim of producing a perspectives paper in the ICES JMS or Fish and Fisheries.

WGECO will report by XXX for the attention of the Advisory Committee.

Supporting Information

Priority	The current activities of this Group will enable ICES to respond to advice requests from member countries. Consequently these activities are considered to have a very high priority. It will also lead ICES into issues related to the ecosystem affects of fisheries, especially with regard to the application of the Precautionary Approach. Consequently, these activities are considered to have a very high priority.

Scientific justification	Term of Reference a) Many stocks are rebuilding and will likely have higher abundance and biomass than we have seen in recent times. This in turn will likely have effects through trophic interactions both up and down the foodweb. At ICES, WGECO and WGSAM have been tasked previously with similar ToRs. WGECO will investigate the potential consequences of stock recovery of benthivorous fish and invertebrates, their ensuing risks for fish stock management and the use of MSFD indicators. It is hypothesized that a large increase in benthivorous fish will have an impact on benthic productivity and biodiversity. This ToR requires data on the spatial distribution of benthivorous predators, their prey consumption rates and diet composition. It also requires data on the abundance and production of benthic faunal. This ToR links to ToR c. Term of Reference b) Identifying thresholds and limits for ecosystem indicators remains a central challenge for ecosystem based fisheries management. This ToR will examine if MSY targets implemented in the current management regime will lead to acceptable ecosystem status. This ToR aims to identify reference levels for a range of ecosystem indicators with the use of size-based models. This proposed ToR links to WGSAM. Term of Reference c) WGECO has traditionally had a leading role in developing and testing indicators, and their use for provision of advice. The work of this ToR facilitates operationalization of these indicators, by identifying data sources, refining, evaluating their strengths and weaknesses and gaps in indicator availability. Indicators that are evaluated to be promising will be tested. Term of Reference d) The ICES strategic plan will end in 2018. This initiative is to allow WGECO to contribute strongly to the development of future ICES strategy. We intend to seek input across the national and disciplinary range of WGECO members, many of whom are operating at a high level in the field and in the home institutes. We aim to publish this as a perspective paper in one of the key journals, and this will be available to inform future progress for this important and centrally positioned Expert Group.
Resource requirements	The research programmes which provide the main input to this group are already underway, and resources are already committed. The additional resource required to undertake additional activities in the framework of this group is negligible.
Participants	The Group is normally attended by some 20–25 members and guests.
Secretariat facilities	None.
Financial	No financial implications.
Linkages to advisory committees	There are no current direct linkages with the advisory committees.
Linkages to other committees or groups	There is a very close working relationship with the groups of the Fisheries Technology Committee, WGBIRD, BEWG, WGBIODIV and WGSAM.
Linkages to other organizations	OSPAR, HELCOM