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

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Executive Summary

The 2014 meeting of WGECO was held at the ICES HQ in Copenhagen, Denmark from the 8–15 April 2014. The meeting was attended by 20 delegates from 15 countries, and was chaired by Anna Rindorf (Denmark). The work conducted was centred on six Terms of Reference concerning foodweb indicator development, the development of Large Fish Indicators (LFIs), possible consequences of "balanced fishing" regimes, effects of fishing on the seabed, ecosystem effects of a landing obligation, ecosystem consequences of rebuilding predatory stocks and two incoming requests from other working groups.

WGECO reviewed the progress on **foodweb indicators**, including the preliminary results of WKFooWI. The group found that indicators tended to fall into two classes, surveillance indicators and indicators for direct management action. Surveillance indicators play an important role in assessment and management of foodwebs, where relations between pressures, drivers, state, and function can be complex and indirect. Further, in line with the results of WKFooWI, WGECO continued work on indicators of functional groups (guilds) and continued the development and testing of foodweb indicators by introducing a new size-based indicator called "Typical Length". The formulation and interpretation of the LFI was discussed, highlighting a need for conceptual clarification of the role of this indicator as a foodweb indicator. The discussion of foodweb indicators was closed with a note on the need for indicators addressing the role of benthos in the food chain.

The progress in the development of **regional and subregional LFIs** was reviewed by the group. General guidelines were made for the frequently encountered problem that only selected fish species are weighed during surveys, creating a need to estimate weight at length by other means. A total of ten LFIs were reviewed covering the regions North Sea, Celtic Sea, Southern Bay of Biscay, Central-Southern Tyrrhenian Sea, Baltic Sea, Poland EEZ, Kattegat North, Kattegat South, The Sound and Gulf of Cádiz. Of these ten LFIs, four had specific thresholds and reference levels assigned and hence can be considered fully developed. An analysis of subregional LFIs in the North Sea showed that the temporal development in larger scale regional indicators is not necessarily related to the development of subregional indicators, and hence regional indicator results cannot be derived from subregional results and vice versa. To ensure that the LFI is above reference levels in all subregions, the analysis must be conducted by subregion, leading to a trade-off between the number of subregions and data support within each subregion.

WGECO considered **"balanced harvesting"** as the adjustment of exploitation patterns to balance the pressures of all fisheries in an area with the relative productivities of the species and sizes of fish. Size-based and other models used to predict the consequences of contrasted fishing regimes have produced nuanced results: Less selective (including balanced) fishing regimes tend to produce higher yields with lower ecosystem impacts in most studies, but the magnitude of the predicted differences varies. The few empirical studies available provide weak evidence that fishing patterns affect community dynamics and biodiversity; the size of the effects is presumably insufficient for a strong signal to be detected among the noise of the many other factors. Balanced fishing may be difficult to implement, both due to less predictable ecosystem dynamics, and due to the complexity of translating the concept into practical management measures. Though it may be precautionary to avoid too selective

fisheries, "balanced fishing" may be at odds with the EU landing obligation if this obligation results in more selective fishing.

Recent progress in the development on **indicators of Good Environmental Status of the benthic community** was reviewed by WGECO. There are substantial ongoing efforts in the BENTHIS project focused on describing the sensitivity of benthic species to fishing and on providing maps of benthic pressure (www.benthis.eu). Sensitivity of benthic species is linked to ten specific traits (Morphology, Maximum body size, Longevity, Larval development, Egg development, Habitat, Position in the sediment, Feeding mode, Mobility and Bioturbator effects) to allow a general evaluation of the sensitivity of a given species without the need for specific experimental evidence of this species. WGECO considered that in addition to these efforts, an important part of defining GES in benthic communities would be to define desirable states of the benthic ecosystem. The group investigated the potential usefulness of the Ecosystem Services Framework in pursuing this definition, and found that this may provide a useful way forward, in particular in the interaction with stakeholders.

WGECO reviewed the **potential ecosystem consequences of a discard ban** and evaluated the need for further research to elucidate this aspect. A direct consequence of banning discards is the creation of a food shortage for scavenging species. The effect of this shortage depends on the ability of the scavengers to compensate by switching to other food sources. Switching to other prey may limit the direct effects on these species, but may cause cascading effects on other species through increased predation or competition. Of the current STECF discard estimates in EU waters, more than half are roundfish and hence consumable by scavenging seabirds. Hence, seabirds are likely to exhibit the first observable effects of a landing obligation, while changes in the scavenging demersal community will be more difficult to observe. In addition, changes in the distribution and selectivity of the fishery impacts the effect on the ecosystem. Given that the extent and direction of changes in the distribution, gear use and selectivity of the fishery are unknown at present and that the knowledge of the potential prey substitution of scavengers is limited, the expected effects of a landings obligation on the ecosystem were described in very broad terms, highlighting the need for further research on scavenging communities.

Concerns about the **potential indirect effects of rebuilding stocks** of piscivorous fish have existed for several decades. WGECO considers that there is some support for the hypothesis of top–down control by predatory fish on prey fish, as prey species generally increase when their predators decline. Several processes may contribute to or modify this response. As predator stocks rebuild, they may become increasingly food limited and prey populations may vary due to other factors independently of predation levels. In cases where prey species decline with increased predation, they can often be maintained above precautionary levels if fishing mortality on prey species is conditioned (reduced) on predator abundance or natural mortality. The combined evidence demonstrates that we should not always expect predators to regulate their prey populations. Further, there is likely to be a substantial bias in the published literature, as lack of correlation is rarely reported in published manuscripts. The effects of rebuilding piscivorous fish species on competing predators such as dependent seabird and marine mammal populations are difficult to predict. As the number of links between piscivorous fish and dependent predators increases, even the sign of the response may be unknown. Concerns about the indirect effects of rebuilding stocks of piscivorous fish therefore do not provide compelling arguments for delaying rebuilding plans.

The ecosystem effects of the recent **increase in plaice biomass in the North Sea** was investigated together with the change in other species with a similar ecological niche. Eight demersal benthivores (plaice, common dab, lemon sole, flounder, grey gurnard, lesser spotted dogfish, cuckoo ray, bullrout) have all been increasing recently, and the increase seemed to be general throughout the North Sea. Together, their increase has caused an increase in predation on benthos of at least a factor 3. Mortality caused by plaice predation alone is assumed to have increased ninefold the most recent years, while fishing-induced benthos mortality has decreased by 10–90%. The absolute change in benthos mortality depends on how the natural mortality of benthos compares to the fishing mortality. More work is therefore required to properly assess the effects of fisheries management on the benthic community and determine if the reduced fishing-induced benthos mortality is offset by the indirect effects caused by an increased mortality of benthos by plaice and other benthivores.

Based on a request from WGISUR for **advice on survey sampling** in the context of ecosystem processes, WGECO recommended a prioritized suite of sampling. The highest priority element focused on improving benthic sampling in the context of the process linking fishing effort to the health of the benthic ecosystem. It was recognized that this would probably require additional sampling effort. The second priority element proposed using routine trawl catches to provide more detailed information on the biology of abundant but non-commercial fish species, extending standard commercial fish sampling protocols to all abundant fish species. A third element was improving data collection related to zooplankton ecology, and in particular to complement CPR and coastal station sampling. Finally, WGECO proposed linking with ecosystem modellers to identify data weaknesses in their models that could be filled by RV surveys. This could also include the development of "testable hypotheses" from the models, which could be empirically evaluated during these surveys.

In conclusion, WGECO noted the need to enhance the development of be**nthic GES indicators** through a list of specific attention areas. Enhancing effort in these areas should provide an increased understanding of the **effect of a landing obligation on the benthic community**, particularly scavengers, as well as an understanding of recent changes in benthic fish communities. More attention is required to determine the **effects of rebuilding predator stocks and changing the distribution of fishing mortality across species and sizes** according to their productivity (increasing "balanced fishing" efforts). The **development of indicators of distribution of species** has received little attention in previous years and is recommended as a priority area of investigation for the future. Finally, WGECO considers that the identification of data needs and **recommendations for further sampling** should be an integral part of considerations for new indicators.

1 Opening of the meeting

The **Working Group on the Ecosystem Effects of Fishing Activities (WGECO)** met at ICES, Denmark from 8–15 April 2014. The list of participants and contact details are given in Annex 1. The chair, Anna Rindorf (Denmark) welcomed the participants and highlighted the variety of ToRs. The draft agenda was presented (Annex 2) and Terms of Reference for the meeting (see Section 2) were discussed. A plan of action was adopted with individuals providing presentations on particular issues and allocated separate tasks to begin work on all ToRs.

2   Terms of Reference

The **Working Group on the Ecosystem Effects of Fishing Activities** (WGECO), chaired by Anna Rindorf (Denmark), will meet in Copenhagen, Denmark 8–15 April 2014 to:

a) Continue the development of foodweb indicators and comment on the suggested foodweb indicators from WKFooWI and WGSAM;

b) Continue work on the large fish indicator, especially in waters other than the North Sea;

c) Consider the ecosystem consequences of "balanced fishing" regimes;

d) Work towards including new research on reducing effects on the seabed and associated communities of fishing operations and gears, including ghost fishing in ecosystem advice;

e) Recommend priority areas of study to determine the ecosystem consequences of landing obligations/discard bans, including survival associated with releasing fish caught;

f) Review knowledge of the consequences to stocks of prey fish (and other parts of the ecosystem) of restoring / maintaining stocks of predatory fish to MSY and recommend priority areas for study.

In addition, the group will consider the following requests from other groups in the ICES system:

WG	REQUEST
WGNSSK	According to WGNSSK estimates, the North Sea is currently ongoing a plaice outburst without precedent. However, plaice is not included in multispecies models, so the consequences of this outburst on the North Sea ecosystem are unclear and would potentially require additional focus
WGISUR	It is recommended that advice be provided on how to design a survey approach to provide ecosystem "process" data, and on what "process" data would be most appropriate

3   ToR a) Continue the development of foodweb indicators and comment on the suggested foodweb indicators from WKFooWI and WGSAM

The development of foodweb indicators has been a key research area for WGECO in recent years (ICES WGECO 2012, 2013b). Several other ICES working groups have participated in the process through dedicated ToRs (for example in WGFE and WGSAM) and in 2014, an entire workshop has been focusing on the identification of available indicators that can be used to inform assessment and management of marine foodwebs (WKFooWI). This section brings together ongoing work of WGECO on foodweb indicators with that of this recent workshop.

3.1   WKFOOWI in the context of WGECO

WKFooWI met one week prior to the WGECO meeting, and it should be noted that this review of WKFooWI is based on a draft report that has not yet been approved by the whole of that group. Key aspects of the approach and outcomes are summarized here. WGECO does not intend to reprise the work of WKFooWI, or to second guess their conclusions, however, where appropriate, we have made additional comments that may be useful in the context of this ToR and for the use of the Review and Advice Drafting Groups for WKFooWI.

3.1.1   WKFooWI basic approach

WKFooWI suggested the following key elements of a process for choosing indicators:

- The need to have a suite of indicators, and not just the "one" indicator;
- The need to have clear criteria for selecting indicators;
- The need to have clear objectives for why indicators shall be developed and used;
- The need to have clear venues for evaluating, vetting and referencing indicators;
- The need to have clear "clients" who will use the indicators and are asking for them.

In addition, indicators should be sensitive, have a basis in theory and be measurable.

This led to a set of high level indicator evaluation criteria to be applied incorporating the following concepts, largely derived from evaluation criteria proposed by WGECO (ICES WGECO, 2013b):

1) Availability of data;
2) Quality of underlying data;
3) Conceptual, Theoretical basis;
4) Communication;
5) Manageable.

WKFooWI also recognized that there was a need for indicators that addressed the full range of "attributes" of foodwebs. This led to evaluation of additional considerations as follows:

- Relation to other MSFD Descriptors;
- The primary foodweb attribute (structural, functional, resilience);
- The Indicator class (energy flow, network, canary, diversity, size, aggregate);
- The Foodweb Functional group (Phytoplankton, Zooplankton, Benthos, Cephalopods, Fish, Birds, Mammals, Reptiles);
- Integrated indicators i.e. indicators that cover processes or attributes across the whole foodweb.

WKFooWI stated that there is a clear need to establish indicator responses and thresholds if they are to be used to inform management and identified best practice approaches to achieve this. They also recognized that this was often not carried out well in practice.

3.1.2 Evaluated indicators

Indicators were evaluated in the context of three primary foodweb attributes: Function, Structure and Resilience (Table 3.1). It should be noted that some indicators were linked to more than one of these attributes.

Table 3.1. Indicators evaluated by WKFooWI.

ECOSYSTEM FUNCTIONING INDICATORS	ECOSYSTEM RESILIENCE INDICATORS[1]	ECOSYSTEM STRUCTURE INDICATORS
Seabird breeding success	Mean trophic links per species	Guild surplus production models
Productivity (production per unit biomass) of key predators.	Ecological Network Analysis derived indicators (overall mean transfer Efficiency)	Total biomass of small fish
Mean weight at age of predatory fish species from data	Gini-Simpson dietary diversity index	Proportion of Predatory Fish
Total Mortality	Herbivory : detritivory ratio	Pelagic to demersal ratio
Primary production required to support fisheries	Ecological network indices of ecosystem status and change (Ulanowicz)	Biomass of trophic guilds
Productive pelagic habitat index (chlorophyll fronts)	System Omnivory Index	Lifeform-based indicator for the pelagic habitat – (also a function indicator)
Ecosystem Exploitation (fisheries)		Region-specific indicators of abundance & spatial distribution
Community Condition		fish biomass/benthos biomass from models
Mean trophic level of catch		Zooplankton spatial distribution and total biomass
Marine Trophic Index of the community (MTI)		Scavenger biomass
Mean trophic level of the community		Geometric mean abundance of seabirds
Disturbance index		Gini-Simpson diversity index (species dominance) of large & small fish by biomass.
Loss in secondary production index (L index)		Species Richness Index
Cumulative distribution of biomass assessment		Large Fish Indicator LFI
Trophic Balance Index (fishing pattern)		Mean length of surveyed community
Mean transfer efficiency for a given TL or size		Size spectra slope
Finn Cycling Index		Zooplankton Mean Size - Total community biomass index

[1] **It should be noted that with the exception of the first in this list, all these indicators were considered to be appropriate as ecosystem functioning indicators as well as for resilience.**

3.1.3 Evaluation of indicators

WKFooWI then carried out a systematic and quantified evaluation exercise along the lines described above and using the broad evaluation approach described by WGECO (2012, 2013) and WGBIODIV (2013). They then proposed a core series of indicators based on the relative ranks within the major attributes as well as a set of more pragmatic criteria detailed below

- Coverage of all attributes; to ensure, to the extent practicable, that all three main categories of attributes were represented.

- Coverage of all functional groups; to maximize the coverage of all functional groups found within a foodweb. Particularly to include lower trophic level taxa that may have scored lower than more commonly or routinely monitored upper trophic levels.
- Major indicator classes; to ensure the major classes of indicators were represented.
- Current operability; related to data availability, management relevance and existence of thresholds, targets or related reference points.
- Links to other MSFD Descriptors; to ensure including indicators that were unique to this MSFD Descriptor.

WKFooWI also suggested two sets of indicators, one set that may be implemented now and one that holds promise for future development on the basis of these additional criteria.

3.1.4   Final selected suite of Indicators for current use

INDICATOR	RATIONALE
Guild level biomass (and production)	These address structural attributes of foodwebs, and can also serve as a proxy for functioning. Improved specification of MSFD D4 indicator, Production per unit biomass 4.1.1 as well the D4 indicator abundance within range 4.3.1.
Primary Production Required to sustain a fishery	This addresses the functioning attribute of foodwebs. Improved specification of D4 indicator, Production per unit biomass 4.1.1.
Seabird (charismatic megafauna) productivity	This addresses the structural attribute of a foodweb, and may be able to serve as a proxy for resilience or functioning. Improved specification of D4 indicator, Production per unit biomass 4.1.1.
Zooplankton spatial distribution and total biomass	This addresses both structural and functional attributes of foodwebs.
Integrated indicators (mean TL, mean size)	This addresses both structural and resilience attributes of foodwebs.

3.1.5   Selected suite of Indicators for future development

The following indicators or collection of indicators were considered as promising for future development but were in need of further research work or data provision to be operational.

- Ecological Network Analysis;
- Gini-Simpson dietary diversity;
- Condition Indicators;
- Marine Trophic Level;
- Primary producers;
- Zooplankton Indicators.

3.1.6 WGECO observations

INDICATOR	WGECO OBSERVATION
Guild level biomass (and production)	This would definitely be useful as a surveillance indicator[1] for the state of the foodweb and the relative stability of its major components. As an operational indicator, it may be difficult to manage, particularly through fishery measures. Given our current state of knowledge, it may also be difficult to set specific targets for the biomass of particular guilds. If management were possible, it may well end up with a focus on particular species within a guild where fisheries measures might be more effective.
Primary Production Required to sustain a fishery	This would appear to be primarily useful as a surveillance indicator[1]. It is difficult to see how specific management could be exerted. If trophic level of specific groups is not constant, the indicator requires persistent sampling of diet composition. It requires context setting and can be difficult to communicate.
Seabird (charismatic megafauna) productivity	These indicators have already been well documented and used in a range of contexts, and can be considered as operational and suitable for management. In the full version of the WKFooWI report, seabird productivity is directly cited as expressing the "abundance" of forage fish, while it actually probably reflects the "availability" of these fish. These indicators are undoubtedly valuable in themselves, but maybe questionable in terms of "integrating" the foodweb below them.
Zooplankton spatial distribution and total biomass	This would be a surveillance indicator[1], for general ecosystem health and productivity–but would not be manageable.
Integrated indicators (mean TL, mean size)	Again, this is a good surveillance indicator. Like guild level biomass, it may be potentially subject to management that focuses on individual components of the community

[1] See Section 3.2 for a definition of surveillance indicator.

Finally, it is noted that WKFooWI also proposed a range of other indicators that might prove useful with further development but were not currently operational. This is an excellent thing to provide. However, WGECO notes that participants in WKFooWI were explicitly asked to propose operational indicators. As a consequence, it seems likely that while the list of operational indicators will be fairly exhaustive, the list of non-operational indicators is likely to be less complete. Further, this is an area of substantial current development and hence, WGECO proposes that the list of indicators for development should be considered as a partial list which will be supplemented in future.

3.2 The need for surveillance indicators

Generally, the most valuable indicators are those which are operational and appropriate to direct management via a pressure-state relationship. So fish stock biomass would be a state indicator, and if this were too low, it could be could managed by e.g. reducing Fishing Mortality; a pressure indicator. A second category of indicators would be surveillance indicators. These are indicators that quantify neither pressures nor directly affected attributes, but are nevertheless needed for an informed assessment and management of foodwebs. Reasons could be that they represent, directly or indirectly, important drivers beyond the control of management, or because they help tracking impacts of management at intermediate points in the causal chains linking manageable pressures to vulnerable attributes of foodwebs. A key feature of surveillance indicators is that they are unlikely to respond unequivocally to management or support target setting. They would operate more to provide warning of changes that

may impact on our ability to achieve targets in other indicators e.g. fishing objectives such as B or F.

"Zooplankton total biomass" is an indicator of this type proposed by WKFooWI. Essentially, this indicator could tell us if there was sufficient food available to the higher trophic levels to support the fish stocks and the fishery removals in an ecosystem where planktivores are food limited, given some understanding of conversion efficiencies etc. It would be operational, in that CPR or satellite data could be used to provide the data support, foodweb models could be used to establish a threshold level; presumably below which the fishery or stocks would be expected to suffer. But zooplankton biomass is not in itself a sensitive ecosystem attribute (in absence of pressures, zooplankton population can rebuild within weeks), and zooplankton biomass levels are not possible to regulate directly. Depending on the cause of zooplankton decline, management responses might be to mitigate potential underlying pressures on primary production (e.g. acidification, pollution), to adjust fisheries management (in case of trophic cascades), or, if the cause is unmanageable, reduce our expectations of sustainable yield from the fishery. In any case, however, information on zooplankton abundance would be crucial to appropriate management action aiming at other connected aspects of the ecosystem. Most surveillance indicators would operate in this way. They would alert managers to changes that will have wider consequences, and the responses would be either change in the management of the state itself, or adjustments of management objectives. Other surveillance indicators, e.g. ocean temperatures may not have such obvious or clear links to the state of managed components. However, warming waters could encourage fish to move deeper, or further north, or enhance their growth, or increase their metabolic rate or impact on their recruitment. It is useful to know that the temperature is rising even if we cannot be sure of the consequences, so the indicator would essentially be a warning signal that conditions are changing. Our response would depend on how well we understood what that change in pressure would do to our managed ecosystem components.

3.3 Structuring suites of surveillance indicators to interpret functional group dynamic

The foodweb is complex not only in structure but also in function. To monitor the degree to which it is affected by management in a comprehensible way therefore requires us to condense information on foodweb status. This is most appropriately done by dividing the structure and function into compartments which share common structural or functional aspects. For the foodweb, such compartments can be the functional guilds such as fish benthivores, fish planktivores, filter feeding benthos or omnivorous zooplankton. The compartments can be classified as more or less important compartments depending on the services they supply to other compartments. For example, in fauna may provide a service to the foodweb in their reorganization of the sediment while forage fish provide services in the form of food availability to higher trophic levels (see Section 6 for a more thorough use of the ecosystem services concept). A wide age distribution may serve to enhance resilience of the system to annual perturbations in environmental conditions. The attributes within each compartment which ensures these services could be total biomass, productivity, diversity of species, size structure of compartments that either in their energy intake or energy transfer show significant size dependence, extent of the possible habitat and proportion or number of K-strategists. To monitor the status of a particular foodweb, the requirement would be to cover the most important guilds in the area, and within each guild,

evaluate the most important processes affecting the services provided by that guild. Hence, some areas would cover a wide range of guilds while others cover a smaller selection.

The MSFD sets out three criteria to determine good environmental status (GES) in respect of marine foodwebs and suggests appropriate indicators with which to monitor change in status and so track progress towards attaining GES. Criterion 4.3 concerns the "*Abundance/distribution of key trophic groups/species*" and indicator 4.3.1 requires metrics of "*Abundance trends of functionally important selected groups/species*" to monitor change in the functional composition of marine foodwebs. Trophic functional guilds have in the past been applied to fish to support both modelling and empirical studies of marine foodwebs. Greenstreet *et al.* (1997) considered four guilds, pelagic planktivores, pelagic piscivores, demersal benthivores and demersal piscivores, which were used by the European Regional Seas Ecosystem Model (ERSEM) (Baretta *et al.*, 1995; Bryant *et al.*, 1995). More recently these same guilds have been used to explore changes in North Sea foodweb trophic structure between 1973 and 2000 (Heath, 2005). In these studies, fish species were assigned to trophic guilds solely on the basis of their diet as adult fish, but this ignores the fact that many species show ontogenetic development in their diet (Daan, 1973; 1989; Robb, 1981; Hislop *et al.*, 1991; Hislop *et al.*, 1997; Greenstreet *et al.*, 1998; Floeter *et al.*, 2005). Assigning fish to trophic guilds based only adult diet ignores the possibility that smaller individuals of the same species may feed at lower trophic levels and fulfil different trophic functions.

A more recent study acknowledges this flaw and assigns the fish sampled in three surveys carried out in the North Sea to five trophic guild based on the diet-at-length of each species (Greenstreet *et al.*, in preparation a). The four guilds used previously are still used, but a fifth, Demersal Planktivores, is added to take account of the juveniles of many gadoid species (Robb, 1981) and to more comfortably accommodate Norway pout. 117 papers were reviewed, which provided sufficient information to assign each 1 cm length class of 95 species sampled by the three surveys to one of the five trophic guilds. Trophic guild assignment was based on the predominant prey category in the diet by weight, thus fish were considered to be piscivores if fish prey constituted more than 50% of the diet by weight. Here we summarize the results derived from just one of the surveys, the first quarter (Q1) International Bottom-trawl Survey IBTS. As a rule, indicator trends derived from the Q1 IBTS and the third quarter (Q3) IBTS tended to be correlated, but the Q3 Dutch Beam Trawl Survey (DBTS) generally produced indicator trends that differed from both IBTS because the larger-sized gadoid species, pelagic fish and Norway pout were poorly sampled in the Q3 DBTS.

Trends in guild biomass are shown in Figure 3.1. The key species contributing to each guild's biomass are also indicated and a five year moving average is fitted to the total biomass to highlight any underlying trends. Over the 29 year period of the Q1 IBTS there was little evidence of any trend in demersal piscivore biomass, but marked changes in variability were apparent. For the first ten years, demersal piscivore biomass fluctuated around a value of approximately 1650 kg km^{-2} by ±250 kg km^{-2}, but from 1993 onwards, variability increased markedly. For example, between 2006 and 2010 biomass varied by 1800 kg km^{-2}, from 800 kg km^{-2} to 2600 kg km^{-2}. The early part of the time-series was characterized by declining cod biomass and increasing whiting biomass. From 1993 onwards, much of the variability was driven by changes in whiting and haddock biomass. Demersal benthivore biomass showed an increasing trend, suggesting on average a doubling from approximately 600 kg km^{-2} to over 1200 kg

km^{-2} over the 29 year period. The index was also highly variable. Over a two-year period, 1999 to 2001, demersal benthivore biomass increased by a factor of ≈3, from 720 kg km^{-2} to 1930 kg km^{-2}, before falling back to 700 kg km^{-2} in the following three years. This variability was primarily driven by haddock, while the general long-term increase in demersal benthivore biomass was mostly caused by increasing common dab biomass[1]. Norway pout accounted for most of the variation in Demersal Planktivore biomass, which was particularly low between 1986 and 1990, and again between 2003 and 2008. There is perhaps a suggestion that when pelagic planktivore biomass was especially high, demersal planktivore biomass was particularly low, and *vice versa*, but the two time-series were not correlated. Trends in Pelagic Piscivore biomass tended to be much higher in the latter half of the time-series and were driven almost entirely by changes in mackerel biomass. Pelagic Planktivore biomass, primarily driven by variation in herring biomass, shows some indication of cyclical variation with peaks in 1986 to 1990, 2002 to 2006, and in 2011, and troughs in 1990 to 1998 and 2008 to 2009.
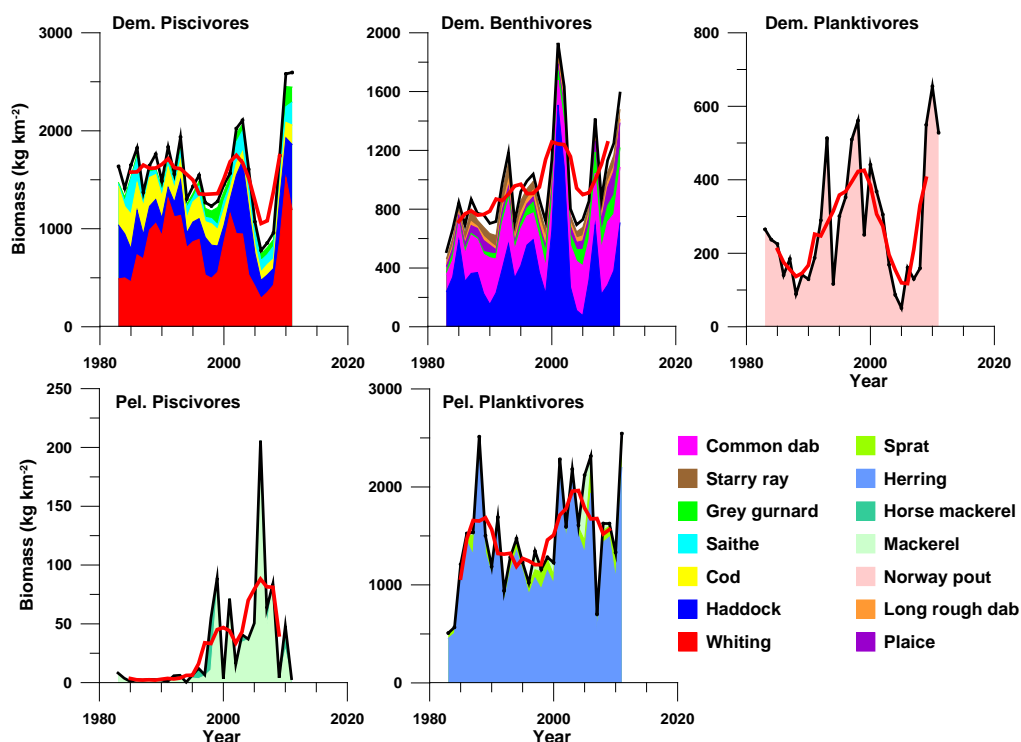
Figure 3.1. Trends in the biomass of Demersal (Dem.) Piscivore, Demersal Benthivore, Demersal Planktivore, Pelagic (Pel.) Piscivore and Pelagic Planktivore trophic guilds taking ontogenetic development in the diet into account derived from the Q1. Contributions to guild biomass by the dominant species in each guild are indicated and total guild biomass is shown by the solid black line. Solid red lines show five year running average smoother on total guild biomass.

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[1] Plaice biomass also increased considerably, but plaice are not so well sampled by the GOV trawl used in the Q1 IBTS, so here it does not play the dominant role in driving the increase in Demersal Benthivore biomass that might be expected. In Section 9.1 we show a similar trend in Demersal Benthivore biomass, but in this instance using Q3 IBTS data corrected to take account of catchability in the GOV trawl. In this example the dominant role in driving the recent increase in Demersal Benthivore biomass is clearly apparent.

Without other information it is difficult to interpret these changes in guild biomass. In the absence of more historic data, these relatively recent changes cannot easily be put into a fisheries disturbance context. Fisheries are thought to have caused declines in piscivorous fish (Christensen *et al.*, 2003; Myers and Worm, 2003), yet the data presented here show no indication of any systematic trend in Demersal Piscivore biomass. The North Sea has been subject to at least a century of heavy fishing pressure (Thurstan *et al.*, 2010), which may have peaked in the mid-1980s, since when fishing pressure on the community may have declined by 50% or more (Greenstreet *et al.*, 2011). The Q1 IBTS time-series therefore mainly covers a period of recovery. Perhaps the Q1 IBTS does not extend back far enough to capture the fishing disturbance impact on the trophic structure of the fish community, and the recovery response of demersal piscivorous fish is too slow to have really established yet (Molloy *et al.*, 2009; Greenstreet *et al.*, 2011; Shephard *et al.*, 2011). A clear increasing trend in demersal benthivorous fish is indicated. This may be precisely the sort of recovery response expected following a marked reduction in fishing pressure, but is this increase in just one of the trophic functional guilds a good thing? Does this represent a North Sea foodweb moving towards GES? To attempt to address these issues, other indicators were applied to examine changes taking place within each of the trophic functional guilds.

Mean asymptotic length was variable over the 29 year period, but the data suggest a sharp decline around 2001 (Figure 3.2); the guild became more dominated by fish with shorter asymptotic length. However, the opposite pattern was suggested in the mean length scaled by asymptotic length metric, a metric of mean length within each population; although the guild was more dominated by species with shorter asymptotic length, the fish making up these population tended to be larger and closer to their asymptotic length. As a result of these two patterns, the mean length and mean weight of fish, while variable, showed no obvious trend. Not surprisingly, overall production was closely linked to changes in guild biomass (compare with Figure 3.1); the influence of the strong 1999 haddock year class is clearly defined, particularly in somatic production. However, productivity (the production/biomass ratio) declined between 2002 and 2003 and then remained at a lower level, and this was driven primarily by a reduction in somatic productivity. The proportion of production directed towards gamete production increased at this time. Species richness of the Demersal Benthivore guild has increased steadily from around 1990 onwards. For the first part of the time-series, species diversity increased, and then declined sharply between 1998 and 2002, before increasing once again.

Figure 3.2. Temporal trends in nine guild composition descriptors applied to the Demersal Benthivore guild. Red lines show five year running means. Black lines show abundance-weighted and grey lines biomass-weighted mean values. Solid lines show geometric mean values and dashed lines show arithmetic mean values.

The nine metrics suggest clear trends in the composition of the Demersal Piscivore guild (Figure 3.3). Firstly, mean asymptotic length declined sharply at the start of the time-series followed by two separate periods of slow recovery. Length scaled by asymptotic length showed a similar pattern of change. Both the relative abundance of large-bodied species and the mean size of fish within individual species' populations initially declined before slowly recovering. Both processes markedly affected the mean size of fish making up the guild; both mean weight and mean length showed marked declines in the first ten years of the time-series followed by very gradual recoveries. By 2011 both mean weight and mean length remained considerably lower than in 1983. Again as anticipated, overall production closely resembled the trend in guild biomass (see Figure 3.1), but productivity varied with a pattern converse to the trend shown by length scaled by asymptotic length. Productivity increased in the first ten years then gradually declined, although productivity was still higher in 2011 than in 1983. This trend was primarily driven by variation in somatic growth productivity and the proportion of production directed towards gamete production showed the reverse trend, declining in the first ten years, then slowly increasing again. No real trend in species richness was apparent, but variation in species diversity closely resembled the trend in mean asymptotic length; as mean asymptotic length initially declined and then showed two subsequent recovery phases, so Hill's N2 initially declined then recovered in two separate stages.

Figure 3.3. Temporal trends in nine guild composition descriptors applied to the Demersal Pis-civore guild. Red lines show five year running means. Black lines show abundance-weighted and grey lines biomass-weighted mean values. Solid lines show geometric mean values and dashed lines show arithmetic mean values.

The Demersal Planktivore guild was almost entirely dominated by one single species, Norway pout (see Figure 3.1), as evidenced by the Hill's N2 species diversity trend (Figure 3.4). Nevertheless, an increasing trend in species richness was apparent, suggesting that over time an increasing number of the early stage small planktivorous size classes of fish that would ultimately become benthivorous or piscivorous were being sampled, but in such small numbers as to have minimal impact on the Hill's N2 trend. Because of the dominance of this single species, mean asymptotic length, mean weight, mean length and mean length scaled by asymptotic length simply reflected the principally environmentally driven recruitment variability within the Norway pout population. Total production by the guild reflected the trend in guild biomass (see Figure 3.1) and productivity and the proportion of production directed towards gamete production varied with no obvious trend through the 29 year survey period.

Figure 3.4. Temporal trends in nine guild composition descriptors applied to the Demersal Planktivore guild. Red lines show five year running means. Black lines show abundance-weighted and grey lines biomass-weighted mean values. Solid lines show geometric mean values and dashed lines show arithmetic mean values.

The Pelagic Piscivore guild mainly consisted of two species, mackerel and horse mackerel (see Figure 3.1), and in 1983 only horse mackerel were sampled, reducing the guild to a single species. In two years salmon were sampled, increasing the species count to three (Figure 3.5). Hill's N2 suggests a decline in species diversity reflecting a reduction in horse mackerel abundance and increased dominance of mackerel (see Figure 3.1). Mean asymptotic length showed no trend; peaks in the arithmetic mean were linked to the inclusion of salmon in the sample. Mean weight and mean length of fish in the guild both showed a declining trend, mainly driven by a reduction in mean length scaled by asymptotic length. Individuals in the population(s) of the dominant species were getting shorter and so further from the species asymptotic length. Overall production varied in line with the guild biomass trend (see Figure 3.1). However total productivity showed indication of a decline despite the fact that somatic productivity may have increased. The proportion of productivity directed towards gamete production showed a marked fall between 1998 and 2008. The data suggest an increasing preponderance of immature fish in the dominant mackerel population.
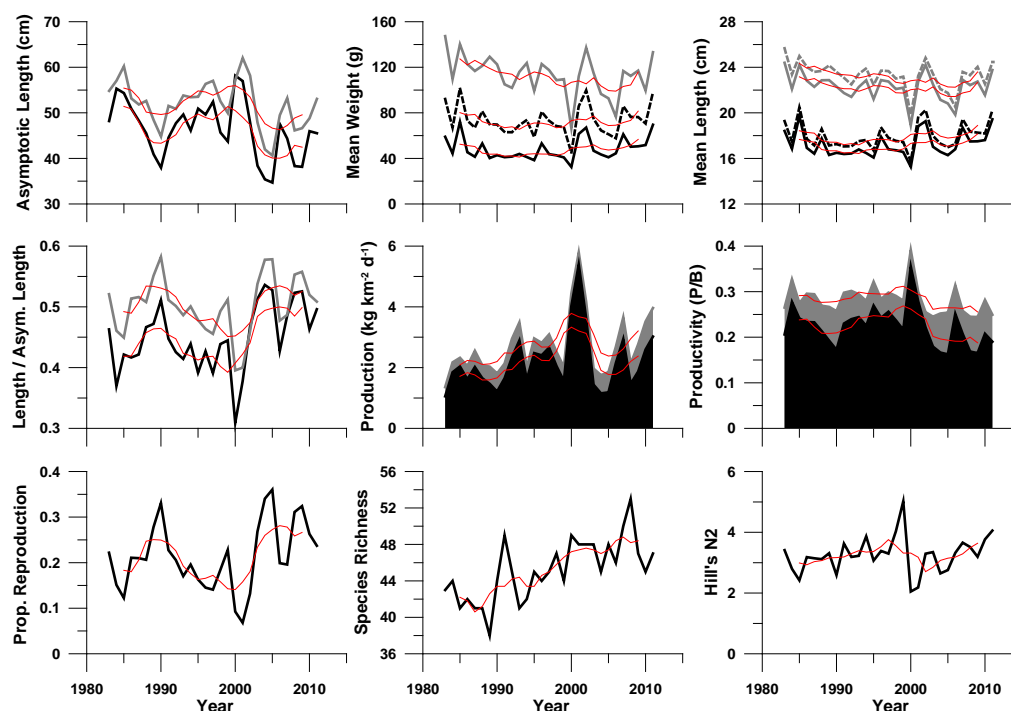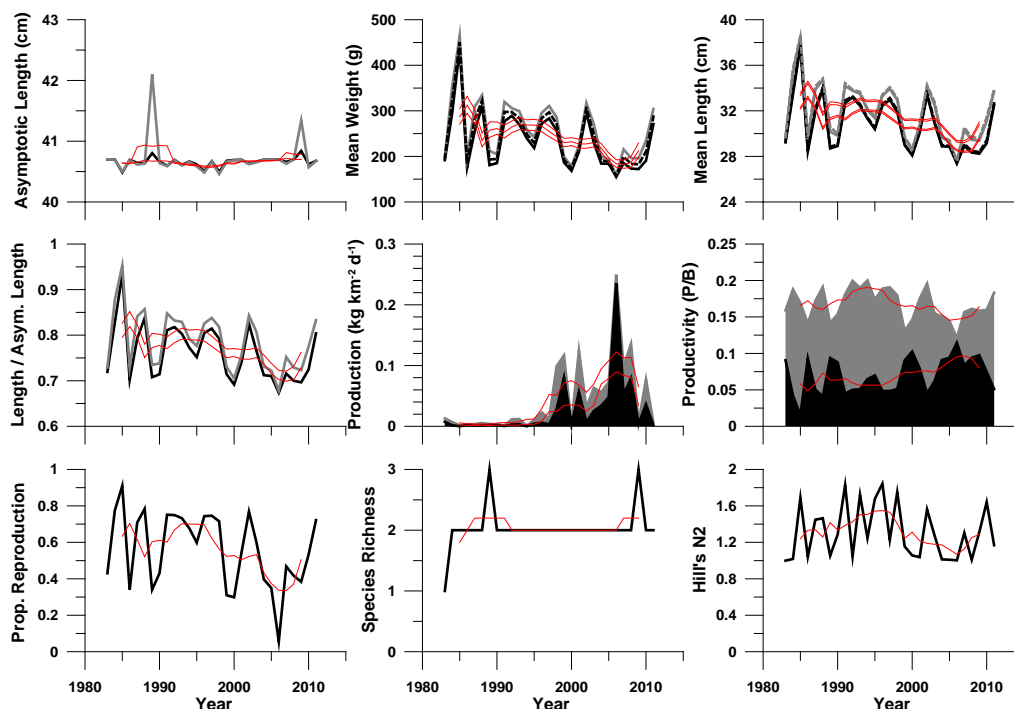
Figure 3.5. Temporal trends in nine guild composition descriptors applied to the Pelagic Piscivore guild. Red lines show five year running means. Black lines show abundance-weighted and grey lines biomass-weighted mean values. Solid lines show geometric mean values and dashed lines show arithmetic mean values.

Species richness of the Pelagic Planktivore guild seems to have increased between 1989 and 1998 and Hill's N2 suggest an increase in species diversity over the course of the 29 year period (Figure 3.6). Nevertheless Hill's N2 remained below a value of two for the majority of the time-series reflecting the dominance of the guild by just two species, herring and sprats, especially the former (see Figure 3.1). Choice of weighting in calculating mean asymptotic length affected interpretation of the metric; when weighted by numbers mean asymptotic length shows a declining trend over the 29 years, but when weighted by biomass no trend is apparent. This weighting issue persists into interpretation of trends in mean weight and mean length. When weighted by biomass neither metric suggests any real trend, but when weighted by numbers both suggest a decline in the size of fish in the guild. If mean size has declined than this is directly linked to reduced representation of larger-bodied species in the guild as no trend in mean length scaled by asymptotic length is apparent. Overall production varied in line with changes in the biomass of the guild (see Figure 3.1) and little trend in productivity is apparent. Similarly, little trend in the proportion of production directed towards gamete production, although this metric is quite variable reflecting the fact that population dynamics of pelagic planktivorous species are strongly influenced by recruitment variability.
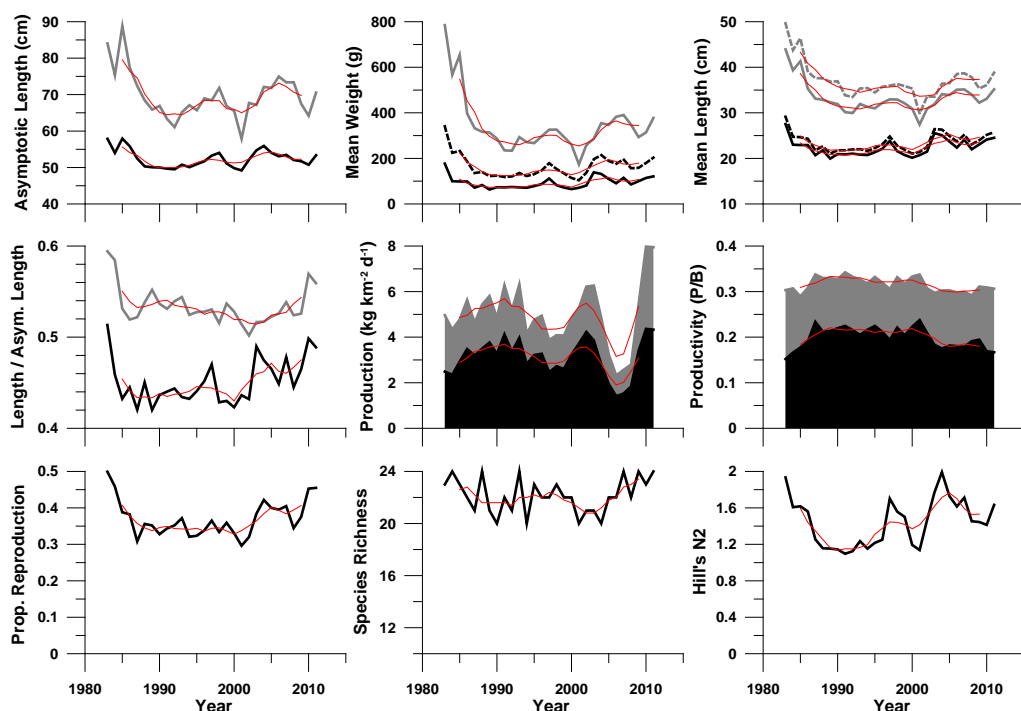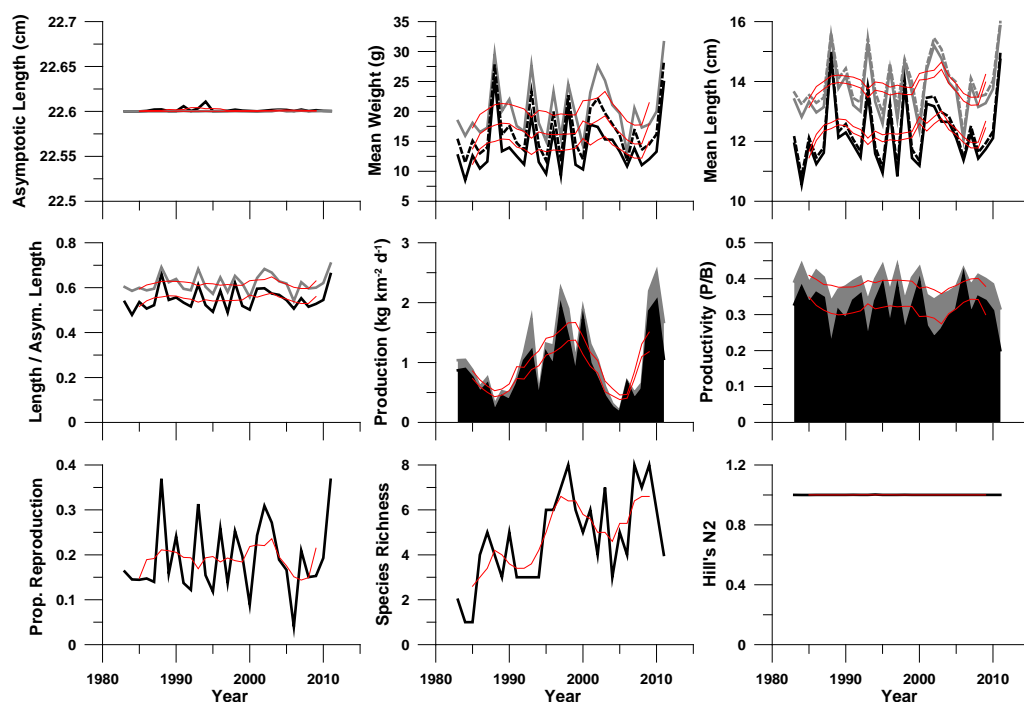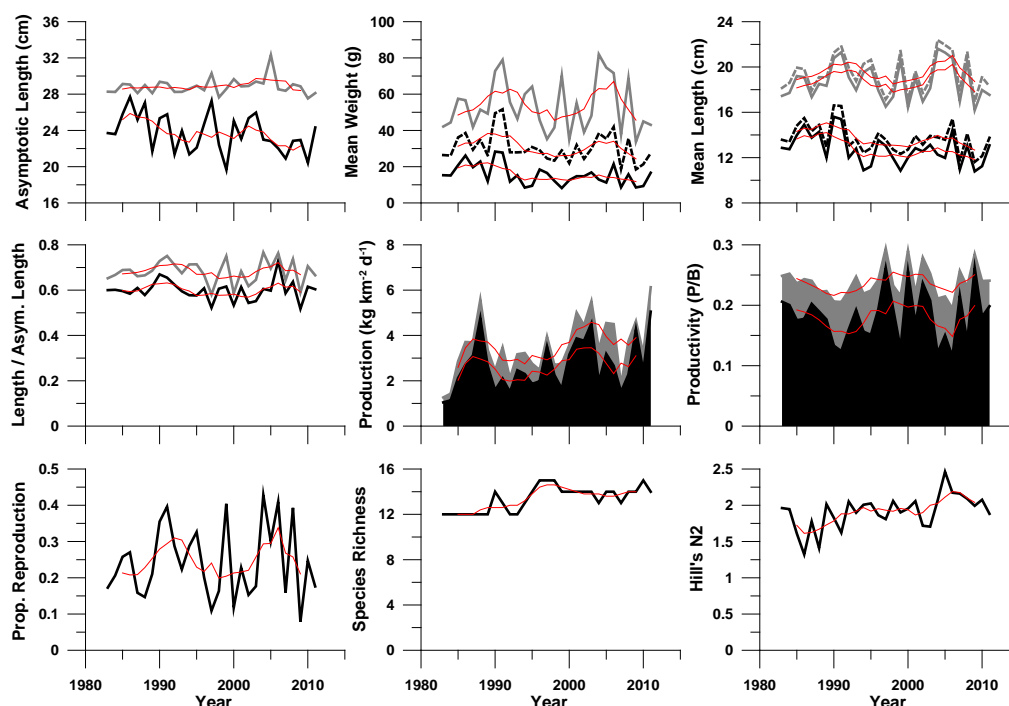
Figure 3.6. Temporal trends in nine guild composition descriptors applied to the Pelagic Planktivore guild. Red lines show five year running means. Black lines show abundance-weighted and grey lines biomass-weighted mean values. Solid lines show geometric mean values and dashed lines show arithmetic mean values.

Review of the nine-metric suites for each trophic guild perhaps suggests that they are more informative for the two main demersal trophic guilds, the Demersal Benthivores and Demersal Piscivores. One possible explanation for this is that the Q1 IBTS is, after all, a demersal trawl survey, and that therefore the demersal guilds are better sampled; metric suites applied to data for these guilds therefore more informative. Countering this argument, Shephard *et al*. (2014) recently applied a suite of indicators to monitor variation in the state of pelagic fish communities in two MSFD Subregion, the Celtic Seas and The Greater North Sea. Comparison with our pelagic guild biomass trends, as well as with data for the main constituent species, revealed strong correlations suggesting that data derived from demersal groundfish surveys can provide a useful basis for developing pelagic guild indicators. Nevertheless, it hard to ignore the fact that one of the most abundant pelagic planktivorous species, the lesser sandeel *Ammodytes marinus*, which is also a key prey species for many fish, seabird and marine mammal predators (Tollit *et al*., 1997; Greenstreet *et al*., 1998; Daunt *et al*., 2008; Reilly *et al*., 2014), was rarely sampled in the Q1 IBTS. The two pelagic fish guilds, and the Demersal Planktivore guild, were all strongly dominance orientated, and this might provide a second explanation as to why these metric suites were perhaps less informative when applied to these three guilds. Nevertheless, knowing about changes in the size and productivity of fish in these guilds does provide additional insight regarding changes in foodweb functioning than knowledge of guild biomass alone can provide.

Changes among this suite of basic metrics can suggest the development and application of more "directed" metrics, which could provide further specific insight as to how foodweb functioning might be changing. For example, the sharp reduction in the relative abundance of large-bodied species, and associated reduction in the mean

length of fish in the Demersal Piscivore guild was one of the main messages to emerge from Figure 3.3. Body size plays a key role in determining trophic function in marine foodwebs (Kerr and Dickie, 2001); larger sized organisms consume smaller sized prey, and so tend to operate at higher trophic levels than smaller sized organisms (Jennings *et al.*, 2001; Jennings *et al.*, 2002a; Sheldon *et al.*, 1972). This has focused much attention to examining predator–prey size relationships. Reviewing this literature, particularly focusing on piscivorous fish predators, and taking account of the fact that piscivores "select" larger sized fish prey, while the smaller sized prey that they consume occur in smaller proportions in the diet than they do in the environment, suggest that piscivorous predators tend to select prey that are approximately 40% of their own body length; a predator–prey length ratio of 2.5 (Daan, 1973; Hislop, *et al.*, 1991; Greenstreet *et al.*, 1998; Scharf *et al.*, 2000; Floeter and Temming, 2005; Reilly *et al.*, 2014).

Assuming a predator–prey size ration of 2.5, knowing the frequency distribution of Demersal Piscivore biomass across each 1 cm length class, and further assuming the predator daily food consumption rates expressed as a proportion of predator body mass shown in Section 3.4.2, the size range of prey consumed daily by the demersal piscivorous fish in 1983, when mean length of fish in the guild was longest, and in 2001, when mean length was least, can be estimated (Figure 3.7). The size of fish prey consumed in 2001 was markedly smaller than in 1983, but the most telling diagnostic is the 95% prey length. In 1983, 95% of fish prey consumed were 43 cm or less, while in 2001 this was reduced to 24 cm. In 2001, once potential fish prey had achieved a length of 24 cm the level of top down control exerted through natural predation mortality was markedly reduced, while to escape to escape top down control to the same extent in 1983 a fish would have to grow to 43 cm. In 2001 <5% of all fish prey consumed would be >24 cm, while in 1984, 24% of fish prey consumed were >24 cm. Considering the growth rates of many potential prey fish belonging to species capable of growing to a length of >43 cm, such a shift in prey size consumption is equivalent to a reduction in the "predation risk time window" of perhaps one to two years. Such a change might well have helped facilitate the rapid increase in the plaice population as alternative top down control from fisheries has been reduced (see Section 3.4.2).
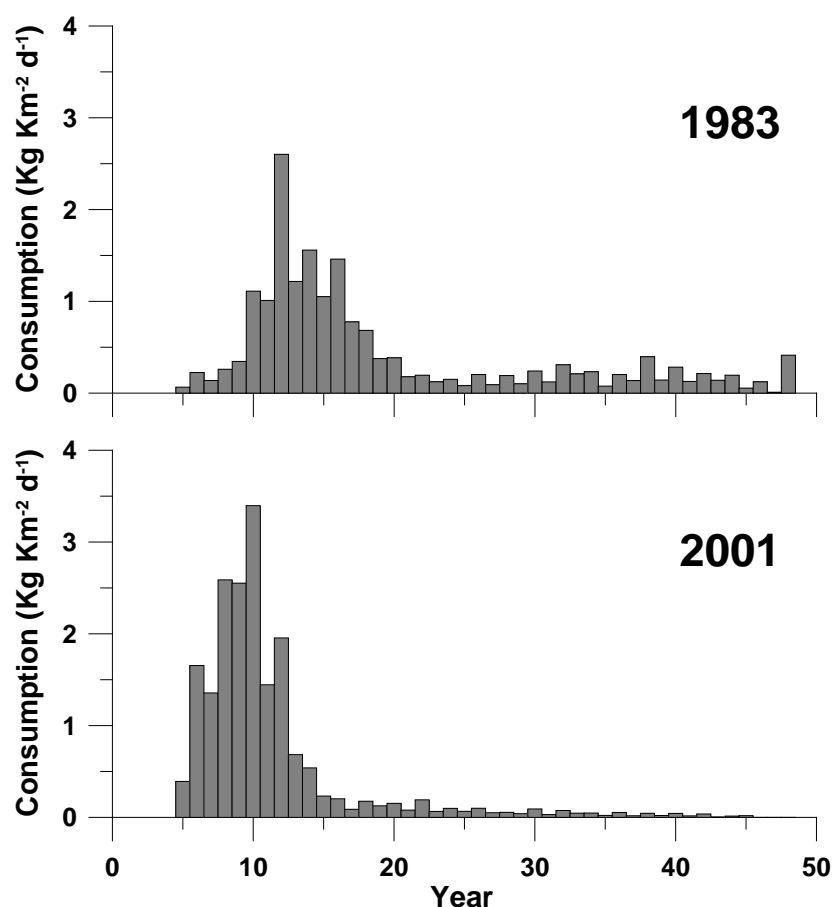
Figure 3.7. Frequency distributions of the size of fish prey consumed in 1983 and 2001.

A further interesting observation is that total daily consumption in 2001 was 18.87 kg km^{-1} d^{-1}, and increase of 4% over the value of 18.22 kg km^{-1} d^{-1} estimated in 1983, despite the fact that in 1983, the biomass of the Demersal Piscivore guild was 5% greater than in 2001. This is because smaller size fish tend to have higher daily food consumption rates expressed as a percentage of body mass than larger sized fish.

3.4 Indicator development and testing

3.4.1 Introduction

In this section we report on advances by WGECO in the development of specific foodweb indicators. Section 3.4.2 addresses the formulations of Criterion 4.2 and Indicator 4.2.1 of the Commission Decision (European Commission 2010), highlighting a need for conceptual clarification. Section 3.4.3 introduces a new size-based indicator with some favourable properties compared with the Large Fish Indicator. Section 3.4.4 reports on work to develop a characterization of the size distribution of marine species through an indicator. Section 3.4.5, finally, is a brief note on the need for foodweb indicators related to benthos.

3.4.2 Examining the relationship between LFI and the proportion of fish at the top of the foodweb in the North Sea

The MSFD sets out three criteria to determine good environmental status (GES) in respect of marine foodwebs and suggests appropriate indicators with which to moni-

tor change in status and so track progress towards attaining GES. Criterion 4.2 concerns the "*Proportion of selected species at the top of foodwebs*", and has as its supporting indicator 4.2.1 "*Large fish (by weight)*". By explicitly stipulating "*large fish*", this criterion considers only the fish components of marine foodwebs. Intuitively, the large fish indicator (LFI), which monitors change in the proportion (by weight) of fish above a specified length threshold that defines "*large fish*" (Greenstreet *et al.*, 2011; Shephard *et al.*, 2011), is precisely the metric necessary to meet indicator 4.2.1 needs. However, the LFI does exactly what its name implies; it monitors change in the size composition of fish communities. The LFI does not explicitly monitor change in the trophic composition of fish communities, i.e. the proportion of the community consisting of piscivorous fish, which might be considered to be "*at the top of foodwebs*". In this section we use a case study based on first quarter (Q1) International Bottom-trawl Survey (IBTS) data, collected across most of the Greater North Sea MSFD Subregion, to examine to what extent and in which sense LFI and proportion of fish at the top of the foodweb correspond to each other.

The LFI has its origins in the study of marine size spectra, i.e. the size distribution of individuals in marine communities (ICES WGECO, 2001, Section 5.3.3.1.2; ICES WGECO, 2005, Section 6.2.3). There is broad agreement that the high regularity of this distribution, observed over many orders of magnitude in body size ("from bacteria to whales", Sheldon *et al.*, 1972) is a result of feeding interactions between different-sized species. The regularity of marine spectra is an emergent property of foodwebs. Most models for aquatic size spectra assume a strict relationship between the trophic level and the size of species, implying that the species at the top of the foodweb are inevitably the largest species. Changes in the proportion of top predators within fish communities might therefore be inferred from changes in the LFI. However, it is well known that this is just a convenient simplification (Jennings *et al.*, 2002a). While there is a correlation between trophic level and logarithmic size, the relation is not perfect. The linkage between fish community size and trophic composition might therefore not be as close as size-based aquatic foodweb theory suggests (Cury *et al.*, 2005). For example, in a heavily fished region of the northwestern North Sea, the expected impact on size composition was observed, but the anticipated coincidental impact on trophic composition was not. Large bodied high trophic level predators were apparently replaced by smaller bodied predators feeding at the same trophic level (Jennings *et al.*, 2002b); size composition changed but trophic composition did not.

In this section, indicators introduced in Section 3.3 are used to explore exactly what the LFI tells us about changes in the trophic composition of the demersal fish community in the North Sea foodweb, and what it does not. And this raises the first important point; as currently defined to support the North Sea EcoQO, the LFI is an indicator of the state of the demersal fish community; the pelagic fish components of the foodweb are not covered by the LFI.

In Section 3.3 we introduced three demersal fish trophic functional guilds: Demersal Piscivores, Demersal Benthivores and Demersal Planktivores. Of the three, Demersal Piscivores feed at the highest trophic level, so a "proportion of piscivores indicator" (PPI) can be determined. The PPI ($I_{PP,y}$) in any one year can therefore be defined as:

$$I_{PP,y} = \frac{B_{Pisc,y}}{B_{Pisc,y} + B_{Benth,y} + B_{Plank,y}}$$

where $B_{Pisc,y}$, $B_{Benth,y}$, and $B_{Plank,y}$ are respectively the biomass densities of Demersal Piscivores, Demersal Benthivores and Demersal Planktivores. Variation in the LFI and the PPI were only weakly correlated; the LFI is not a particularly good indicator of the proportion of piscivores among the demersal fish community. When a similar analysis was performed using third quarter (Q3) IBTS data and Q3 Dutch beam Trawl Data (DBTS) neither of the correlations was significant. The LFI is not a good proxy for the PPI (Figure 3.8).
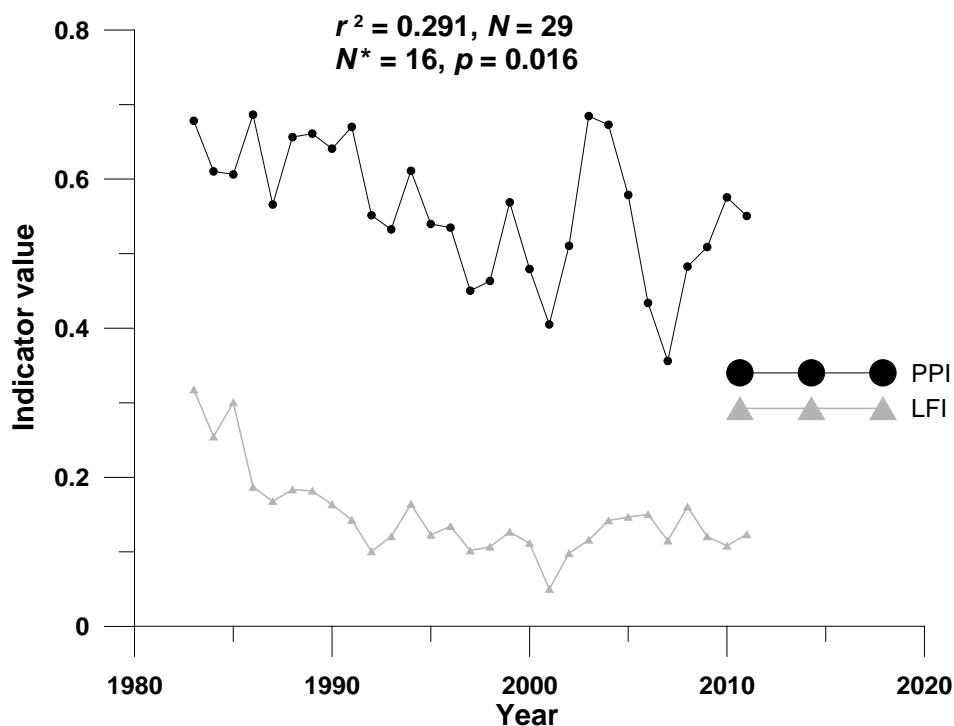
Figure 3.8. Large Fish Indicator (LFI) and Piscivore Proportion Indicator (PPI) time-series trends determined using Quarter 1 International Bottom-trawl Survey (IBTS) data. Squared correlation coefficients comparing the two *time-series* trends are given (r^2), along with the actual sample size (*N*) and the effective sample size (*N**) determined using the modified Chelton procedure. The one-way significance probability (*p*) is shown based on *N**-2 degrees of freedom.

In Section 3.3 we presented the trend in a "length of ontogenetic development of piscivory indicator" (LODPI), the average length at which the fish that make up the Demersal Piscivore guild develop a piscivorous diet. Although resembling the LFI trend, the two indicators were not correlated once autocorrelation within the two time-series was taken into account (Figure 3.9). So the LFI was also not a good proxy for the LODPI.
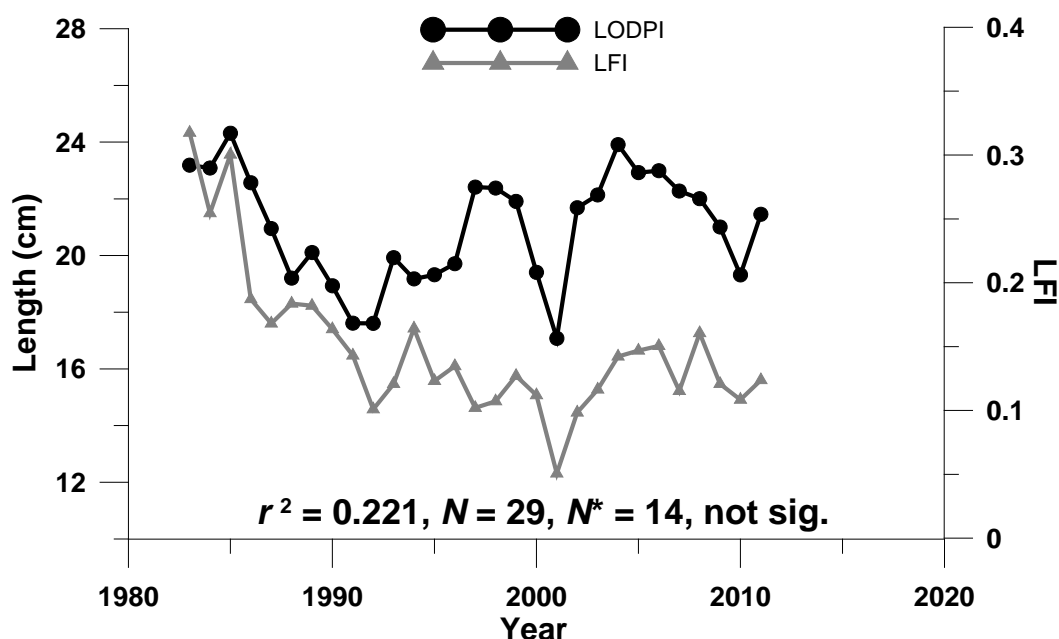
Figure 3.9. Large Fish Indicator (LFI) and Length of Ontogenetic Development Piscivory Indicator (LODPI) time-series trends determined using Quarter 1 International Bottom-trawl Survey (IBTS) data. Squared correlation coefficients comparing the two time-series trends are given (r^2), along with the actual sample size (N) and the effective sample size (N^*) determined using the modified Chelton procedure. The significance probability (p) is shown based on N^*-2 degrees of freedom.

Trends in mean asymptotic length, mean weight, mean length and mean length scaled by asymptotic length of fish in the Demersal Piscivore guild are shown in Section 3.3. The LFI is a size based metric, and since these are all metrics of the size of fish in the guild, it is perhaps no surprise that the LFI was correlated with all five metrics (Figure 3.10). Biomass-weighted mean values of the various metrics were all more strongly correlated with the LFI than abundance-weighted means, which again was to be expected given that the LFI is itself a weight-based indicator. The LFI is a good proxy for the mean size of fish within the Demersal Piscivore guild.

Metrics of productivity of the fish in the Demersal Piscivore guild were also presented in Section 3.3. These were not correlated with the LFI. The LFI is not a good proxy for productivity among demersal piscivorous fish.
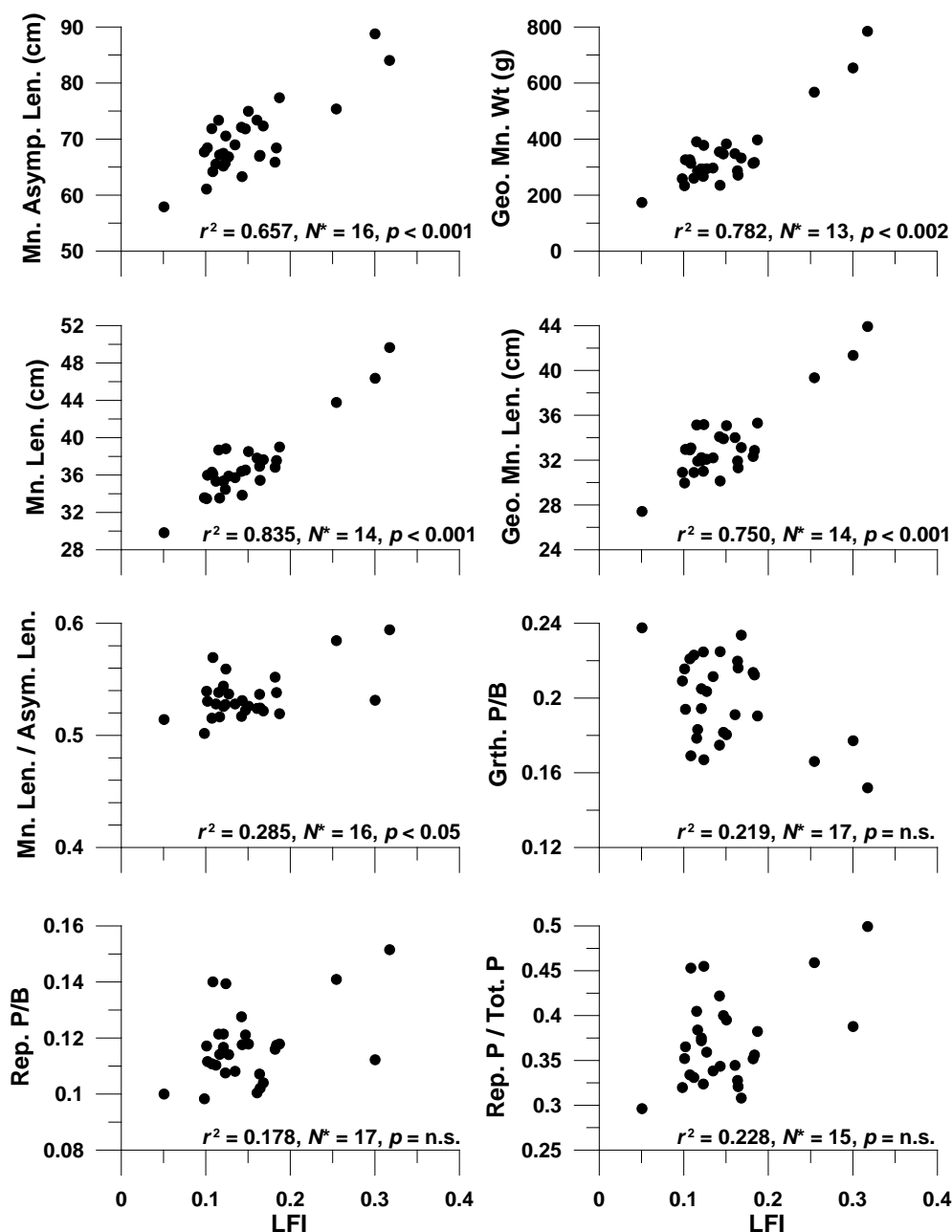
Figure 3.10. Relationships between the LFI and mean asymptotic length, geometric mean weight, mean length, geometric mean length, mean length scaled by asymptotic length, somatic productivity, gametic productivity, and the proportion of total production directed towards gamete production of fish in the Demersal Piscivore guild.

Much of the concern regarding the impact of fishing on marine foodwebs related to the loss of top predators within the fish community (Christensen *et al.*, 2003; Myers and Worm, 2003). In modern times simply being piscivorous might be considered sufficient for a fish to be considered a top predator. However, among marine mammal and seabird components of marine foodwebs, the apex predators are often considered to be those species that feed on fish prey that are themselves piscivorous, such as gannets, harbour seals and grey seals consuming mackerel, whiting and cod (Martin, 1989; Prime and Hammond, 1990; Hammond *et al.*, 1994; Tollit *et al.*, 1997; Hamer *et al.*, 2000). Similarly, it is the loss of the very largest piscivorous fish in the community, those large enough to consume fish prey that are themselves piscivorous,

that could therefore constitute the principal cause for concern. Size-based models suggest that over the course of the 20th century, fishing may have caused a >90% reduction in the abundance of fish of >4 kg body weight in Northeast Atlantic continental shelf seas (Jennings and Blanchard, 2004), an assertion that has some empirical support (Greenstreet and Hall, 1996; Quero, 1998; Casey and Myers, 1998; Rogers and Ellis, 2000; Stevens *et al.*, 2000). A separate subgroup within the demersal piscivore trophic guild was therefore defined, the Demersal Apex Predators; those piscivors whose fish prey were also piscivorous.

Given the predator–prey length ratio of 2.5 for piscivorous fish consuming fish prey established in Section 3.3, fish within the Demersal Piscivore guild were considered to be apex predators at and above body lengths 2.5 times longer than their initial length of ontogenetic development of piscivory. For example whiting develop a piscivorous diet at 13 cm. In the size range 13 cm to 32 cm therefore, whiting were simply Demersal Piscivores, but whiting 33 cm in length and longer were deemed to be Demersal Piscivore Apex Piscivores, capable of consuming fish prey that were themselves piscivorous (e.g. Greenstreet *et al.*, 1998). Figure 3.11 shows trends in both the total biomass density of Demersal Piscivore Apex Piscivores and the proportion of the Demersal Piscivore guild biomass consisting of Apex Predators. The LFI was closely correlated with the latter (r^2 = 0.603, N^* =13, p = 0.0004). Thus, in the North Sea, the LFI is a good proxy for the proportion of apex predators, piscivorous fish that consume piscivorous fish prey, within the Demersal Piscivore guild.
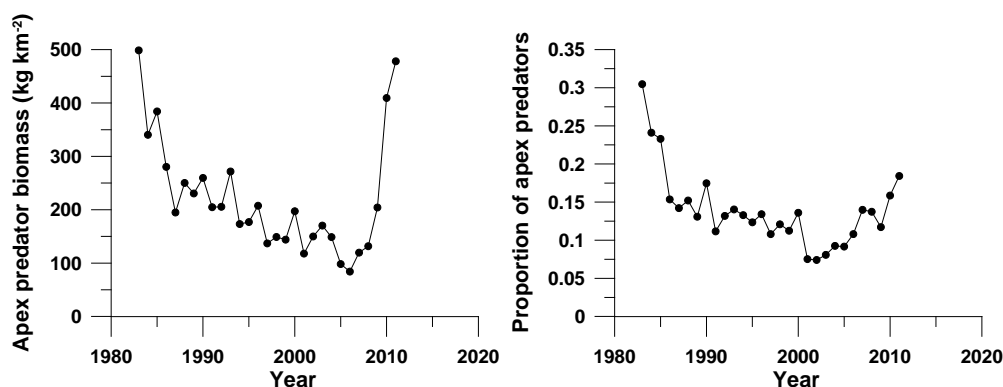
Figure 3.11. Temporal development in apex predator biomass and the proportion of apex predators in the demersal community.

3.4.3   Integrative size based indicators

To conclude, the assumption implied in the formulation of Criterion 4.2, that size and trophic level of fish are interchangeable with each other, is not sufficiently satisfied in natural foodwebs to base guidance for indicator development on it. While indicators based on trophic guilds, such as that of Demersal Piscivore or Demersal Piscivore Apex Piscivores, have just as important a role to play in assessments of the ecological status of foodwebs as size-based indicators, the information they convey is not necessarily the same. Nevertheless, our analysis of the North Sea dataset supports the notion that LFI is related to the proportion of Demersal Piscivore Apex Piscivores, a finding that is relevant to the interpretation of both metrics. Integrative size based indicators

3.4.3.1 Introduction: looking beyond the Large Fish Indicator

Fish community size structure is known to be sensitive to fishing and to require considerable time to recover from anthropogenic perturbations, in particular when the abundance of large fish has disproportionally declined. The Large Fish Indicator (LFI, see also Section 4), defined as the proportion by weight of large fish in survey samples, has been developed by WGECO as an indicator that is sensitive and specific to this vulnerability of fish community size structure. It is now widely calculated throughout Europe (Section 4). The Marine Strategy Framework Directive (MSFD) explicitly specifies it as an indicator for foodweb (D4) GES.

However, in the practice of adopting the LFI as an MSFD indicator at Regional or Subregional level, some complications became evident. It is known that the LFI achieves its sensitivity and specificity only when the threshold length L_{th} used to divide individual fish into the "small" and "large" categories is chosen adequately for a given assessment area (Shephard *et al.*, 2011). In some cases, however, survey time-series are too short or contrasts too small to execute the method for determining an adequate threshold value. Furthermore, if different threshold values are employed for different assessment areas, it becomes difficult to compare LFI values for these areas and to aggregate them for the purpose of larger-scale assessments. Here, we propose an alternative indicator for fish community size structure, the Typical Length, which does not contain a free parameter in its definition and, based on the limited analyses we performed, has statistical properties comparable to those of the LFI.

The Typical Length (TyL) is defined as the geometric mean length of fish, weighted by body mass. In other words, if there are N fish in a sample and M_i and L_i denote body mass and length of the i-th fish, respectively, then:

$$\text{TyL} = \exp\left[\frac{\sum_{i=1}^{N} M_i \ln(L_i)}{\sum_{i=1}^{N} M_i} \right].$$

Below, we first provide an example for a time-series of TyL computed from surveys and compare it with the LFI. We then argue mathematically that TyL and LFI can be expected to have more favourable statistical properties than other size-based indicators that have been proposed, and can therefore be expected to be more informative for assessment and management applications.

3.4.3.2 Comparing Typical Length and Large Fish Indicator in the Celtic Sea

Figure 3.12 displays time-series of TyL and LFI computed for the Celtic Sea, based on the (no longer active) first quarter (Q1) UK West Coast Groundfish Survey (WCGFS) from 1986 to 2004. Calculations draw on methods and R scripts of Shephard *et al.* (2013). The TyL in the period 1986–1990 was around 40cm, and then gradually declined to values around 25–30 cm until 2004. The trend is statistically robust despite considerable confidence intervals, and largely follows the trend of LFI, which was computed with a threshold length of $L_{th} = 50$cm following Shephard *et al.* (2011). Specifically, the correlation between the true time-series of log(TYL) and LFI lies in the rage 0.933 to 0.978 with 95% probability, based on 10 000 pairs of indicator time-series derived from resampled survey data, which were obtained by bootstrapping hauls within years.

To compare the specificity of TyL and LFI to fishing rather than other processes, such as variation in recruitment, we evaluated the trends of both indicators over time. Since fishing pressure over the time period considered was fairly constant (Shephard *et al.*, 2013) and it is known that fish community size structure integrates fishing pressures over long periods of time (Rossberg, 2012; Fung *et al.*, 2013), a gradual, steady decline is the expected response of LFI and TyL to fishing; and the strong observed trends in both indicators are unlikely to have resulted from other drivers. As a measure for the relative specify of log(TyL) and LFI, we therefore computed the difference between sample correlations corr(LFI,year) and corr(log(TyL),year). The analysis was conducted based on log(TyL), because logarithmic length (or similarly logarithmic body mass) plays a conceptually more important role in ecological theory than length itself (Section 3.4.3.3). To separate the effects of actual fluctuations in the indicator values from measurement uncertainty, the difference was computed for 10 000 indicator time-series obtained from bootstrapped survey data resampled as above. Confidence intervals for the difference based on this analysis are shown in 3.13 as a function of Lth. With the literature value for large fish threshold, Lth = 50 cm, there is statistical evidence that LFI is more sensitive to fishing than log(TyL). For other threshold values, the statistical evidence is marginal. Considering that the value Lth = 50 cm had been determined through a similar exercise based on the same dataset (Shephard *et al.*, 2011), one should not over-interpret the higher specificity of LFI for this threshold value.

Similar conclusions can be drawn when applying the simulation method of Houle *et al.* (2012) to compare the specificity of TyL and LFI to fishing pressures (Figure 3.14). For fishing with unselective trawlnets, specificity of TyL and LFI is predicted to be similar for small to intermediate fishing efforts. For high fishing effort with trawlnets, and any level of fishing effort with size-selective gillnets, however, LFI is notably more specific than TyL.

To quantify the sensitivity of the indicators, the signal-to-noise ratio (S/N) can be used when "noise" relates to sampling errors only and the "signal" is the true indicator time-series. Specifically we compute the ratio between the standard deviation of the indicator time-series and the root mean square of the standard deviation of sampling errors, estimated by bootstrapping as above. Based on the observed indicator time-series, S/N is 2.65 for LFI and 2.44 for log(TyL), indicating that LFI is slightly more sensitive than TyL. In order to take into account that the observed indicator time-series themselves contain measurement errors, we evaluated this comparison for 10 000 pairs of indicator time-series computed from bootstrapped survey data, so simulating alternative conceivable outcomes of the survey program. S/N was higher for LFI than for TyL in 89% of cases. Thus, there is an 11% probability that, for the survey program considered, S/N for TyL is higher than for LFI. This comparison, too, might be biased by the fact that Lth had been optimized for this particular dataset. We conclude that, based on analyses presented here, the information provided by LFI and TyL is very similar, and that sensitivity and specificity of LFI and TyL are similar, with potentially a marginally better performance of LFI.

3.4.3.3 Mathematical analysis of indicators for community size structure

One can argue on general theoretical grounds that LFI and TyL should have particularly favourable statistical properties compared with other length-based indicators. The important ecological fact to notice is that the biomass of a community is often distributed over individuals of different sizes in such a way that similar amounts of biomass are allocated to body-size classes spaced evenly on a logarithmic body-size

axis. In other words, if $\lambda = \ln L$ denotes logarithmic length and $B_{>\lambda}$ is the biomass of individuals with logarithmic length larger than λ, then the density of biomass on the logarithmic length axis, given by $D(\lambda) = -dB_{>\lambda}/d\lambda$, does not vary dramatically with λ, up to some point where $D(\lambda)$ drops to zero. When expressing the values of various length-based indicators in term of integrals over $D(\lambda)$, as done in Table 3.1, it becomes clear that for most conceivable indicators the integrand contains an exponential function that either increases or decreases dramatically with λ, so that the integrals are dominated by the contributions from either the largest or the smallest individuals. Fluctuation in the numbers of the largest or the smallest individuals will therefore translate directly to corresponding fluctuations in the indicator values. These fluctuations are known from empirical as well as numerical studies (e.g. ICES WGECO, 2007; Houle *et al.*, 2012).

For LFI and TyL, however, this is not the case. Both indicators give approximately even statistical weight to all logarithmic length classes, and so integrate more effectively the information contained in $D(\lambda)$, while minimizing the impact of abundance fluctuations in particular length classes. Favourable statistical properties of LFI and TyL compared with other length-based indicators are therefore to be expected.

Because the ranges in λ to which populations of individual species contribute are rather narrow (Shephard *et al.*, 2012), these considerations are likely to translate to indicators relating to the size of species rather than the size of individuals, such as Mean Maximum Length, or the Large Species Indicator defined by Shephard *et al.* (2012).

Differences among LFI time-series derived for similarly designed surveys in the same area generally turn out to be small compared to typical sampling errors (e.g. Greenstreet *et al.*, 2011, Figure 3). For TyL, a similar robustness can be expected based on the considerations above. Nevertheless, it must not be forgotten that, just as the LFI, the value of TyL is defined with reference to a particular sampling design, and so will vary depending on design, especially for a varying lower size cut-off of the size range coved by the sampling gear.
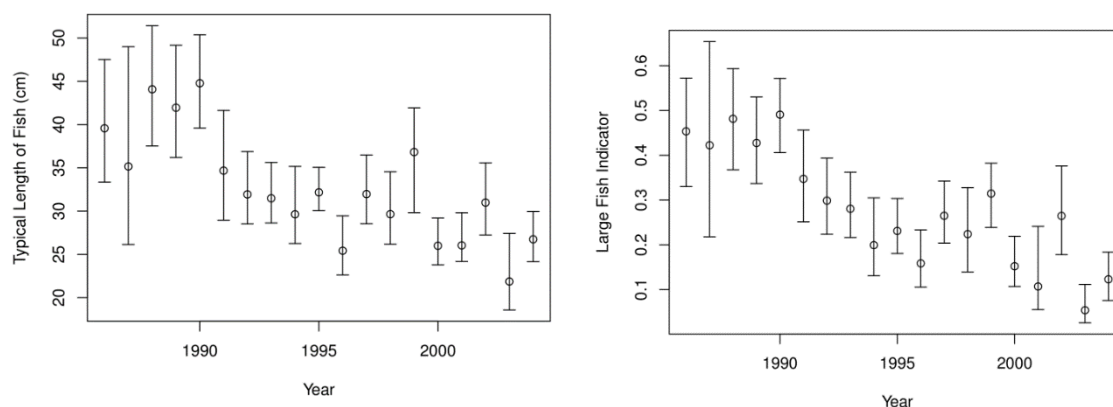
Figure 3.12. Time-series of Typical Length (TyL, left) and Large Fish Indicator (LFI, right) computed for the Celtic Sea's demersal fish community. Vertical bars denote 95% confidence intervals based on bootstrapping of hauls following Shephard *et al.* (2013).
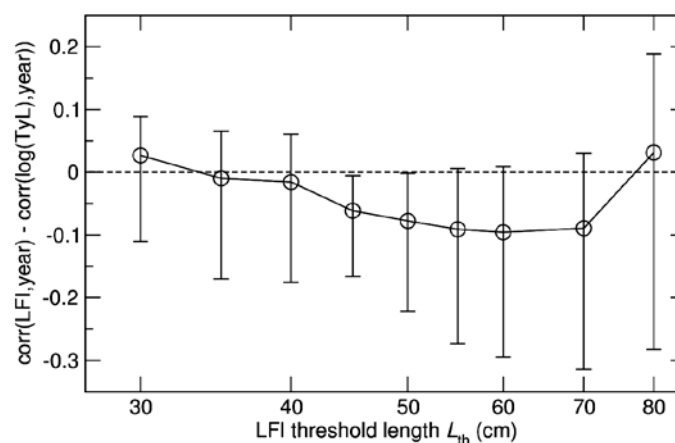
Figure 3.13. Comparison of specificity of TyL and LFI to fishing depending on the value chosen for the large-fish threshold L_{th} (literature value: 50 cm). Negative values indicate LFI is more specific to fishing than TyL. Vertical bars are 95% confidence intervals computed by bootstrapping.
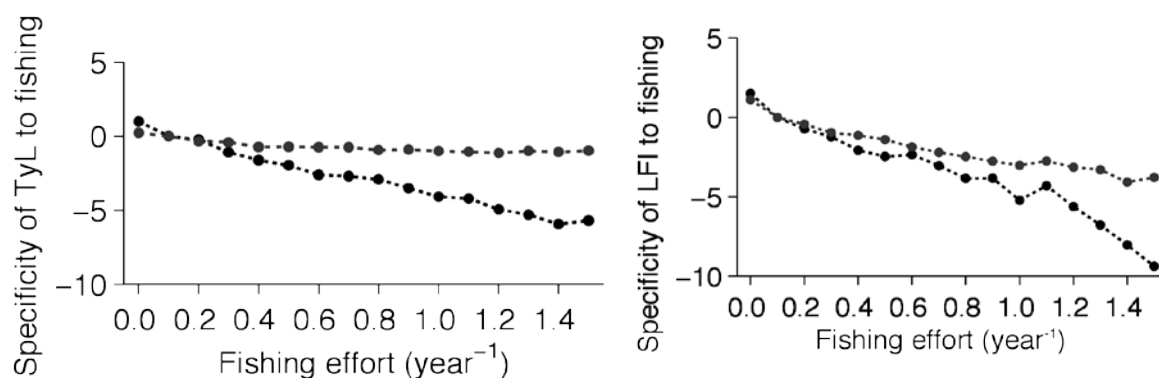
Figure 3.14. Comparison of the simulated specificity of TyL and LFI to fishing using the method of Houle *et al.* (2012). Specificity is here defined as the response to fishing pressure divided by the rms change in the indicator value after small random variations of model parameters without fishing. Grey dotted lines correspond to fishing with size-selective gillnets, black dotted lines to fishing with unselective trawlnets.

3.4.3.4 Conclusions

Typical Length (TyL) exhibits time-series and statistical properties quite similar to the LFI, but has the additional advantages of not requiring determination of a large fish threshold L_{th} for each study area considered. This is particularly advantageous when survey data are insufficient for identifying an appropriate threshold. Because TyL is defined as an average, TyL values computed for smaller study areas are easily aggregated to TyL values at regional or subregional level.

Table 3.1. Representations of size-based indicators through integrals over the distribution $D(\lambda)$ of biomass over the logarithmic length axis. It can be seen that LFI and TyL weight different size classes more evenly than other sizes based indicators. λ_0 is the lower length cut-off of the sampling gear; $H(x)$ is the Heaviside function, which evaluates to 1 for $x > 0$ and to 0 otherwise. Suffix B means statistical weighting by individual body mass, suffix N means weighting by number. In the latter case, body mass is approximated to be proportional to L^3, which leads to the exponents $e^{-3\lambda}$ in the integrands.

$$\mathrm{LFI} = \frac{\int_{\lambda_0}^{\infty} D(\lambda) H(\lambda - \ln L_{\mathrm{th}})\, d\lambda}{\int_{\lambda_0}^{\infty} D(\lambda)\, d\lambda}$$

$$\ln(\mathrm{TyL}) = \frac{\int_{\lambda_0}^{\infty} D(\lambda)\, \lambda\, d\lambda}{\int_{\lambda_0}^{\infty} D(\lambda)\, d\lambda}$$

$$\mathrm{Mean\ Length}_{\mathrm{B}} = \frac{\int_{\lambda_0}^{\infty} D(\lambda)\, e^{\lambda}\, d\lambda}{\int_{\lambda_0}^{\infty} D(\lambda)\, d\lambda}$$

$$\mathrm{Mean\ Length}_{\mathrm{N}} \approx \frac{\int_{\lambda_0}^{\infty} e^{-3\lambda} D(\lambda)\, e^{\lambda}\, d\lambda}{\int_{\lambda_0}^{\infty} e^{-3\lambda} D(\lambda)\, d\lambda} = \frac{\int_{\lambda_0}^{\infty} e^{-2\lambda} D(\lambda)\, d\lambda}{\int_{\lambda_0}^{\infty} e^{-3\lambda} D(\lambda)\, d\lambda}$$

$$\mathrm{Mean\ Weight}_{\mathrm{N}} \approx \frac{\int_{\lambda_0}^{\infty} D(\lambda)\, d\lambda}{\int_{\lambda_0}^{\infty} e^{-3\lambda} D(\lambda)\, d\lambda}$$

3.4.4 Development of an indicator for the species–size distribution

In previous work (ICES WGECO, 2013b, Section 7.2), WGECO noted that the body masses of species found in demersal communities closely follow Pareto (power-law) distributions. Here we present simulation result demonstrating the relevance of this finding for foodweb GES, and report on new analyses with the aim of constructing an indicator to characterize changes in this distribution.

3.4.4.1 Simulations

Figure 3.14 (top) shows the species-size distribution emerging in a typical foodweb generated by the assembly algorithm of the PDMM. In a first numerical experiment, we removed 50% of all primary producers from the foodwebs and simulated population dynamics, in a second experiment we removed the larges eleven species. As can be seen in Figure 3.14 (centre), the first experiment leads to the extinction of about 50% of species at all size classes, so that, ultimately, a Pareto distribution with a slope close to the original value of -0.17, but half of the number of species, is obtained. The removal of top predators in the second experiment, on the other hand, has much

smaller impacts on the richness of species at lower levels. The simulations demonstrate that, in foodwebs, i.e. communities where trophic interactions dominate over other interactions, large species at high trophic levels are highly sensitive to loss of diversity at lower trophic levels.
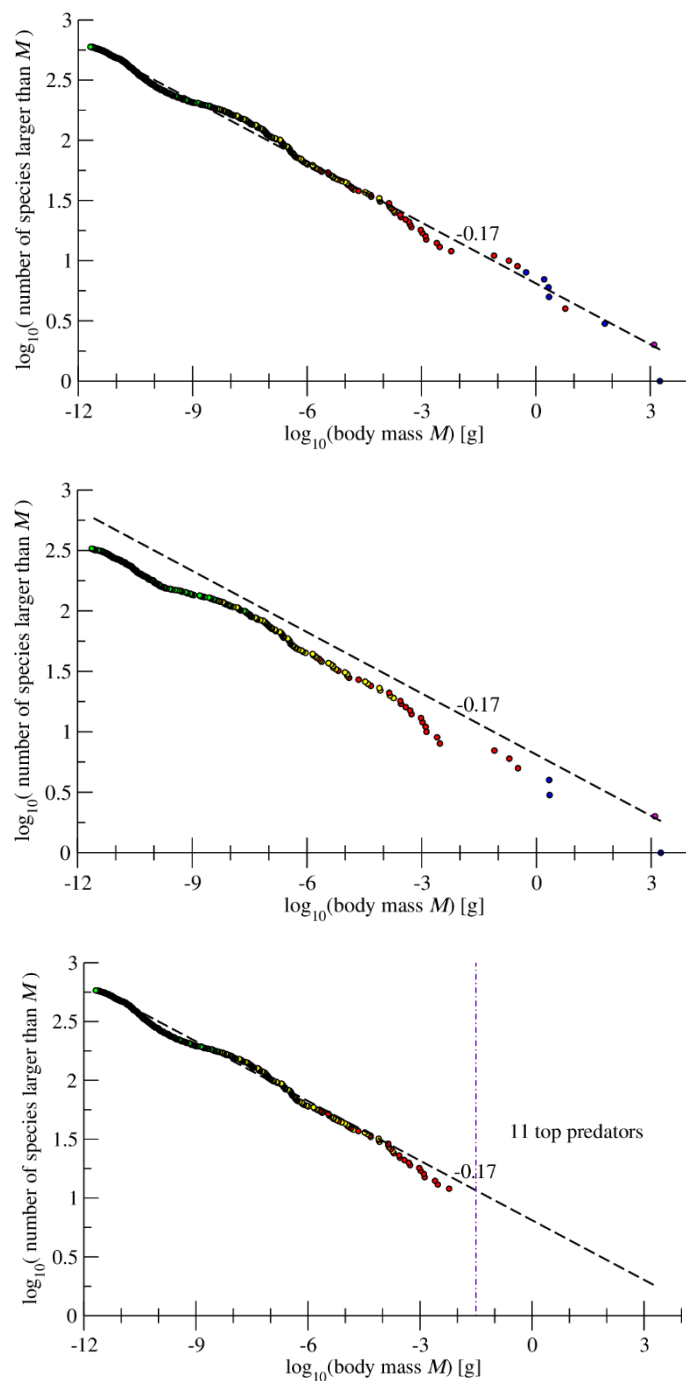
Figure 3.14. Simulations of responses of the species-size distribution to species removal. Top: original foodweb. Centre: foodweb after removal of 50% of primary producer species (tropic level 1). Bottom: foodweb after removal of eleven largest species. Colours code nearest integer trophic level: 1=green, 2=yellow, 3=red, 4=blue.
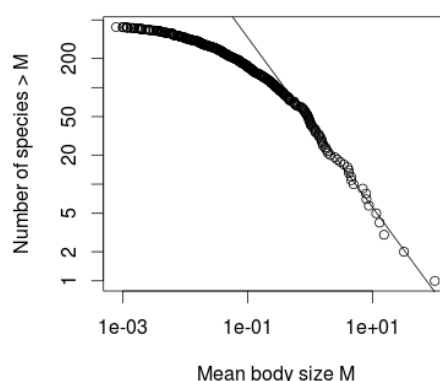
Figure 3.15. Species size distribution (body masses in kg) in the Portuguese Autumn Bottom-trawl Survey, pooling data from 1980 to 2012.

3.4.4.2 Data

New empirical data corroborate the Pareto distributions we found previously, but also highlight challenges with identifying trends in these distributions. We analysed two datasets: (1) body sizes of fish and benthos in the Portuguese Autum Bottom-trawl Survey, kindly provided by Hugo Mendes, IPMA, and (2) species sizes in the Norwegian-Russian ecosystem surveys in the Barents Sea (Anon., 2009; 2010; 2011; Eriksen, 2012), courtesy to Lis Lindahl Jørgensen, IMR.

The body-size range covered by the Portuguese survey is rather narrow (Figure 3.15), so that a clear Pareto distribution cannot emerge. Nevertheless, the data might be of use to identify changes in the richness of species of different sizes through time, and interactions between richness at different sizes as demonstrated in the simulations. To test for such effects, we divided the set of observed species into those smaller and larger than 100 g and computed the time-series of richness for both classes. The time-series (Figure 3.16) are characterized by strong fluctuations, likely due to methodological variations during the sampling period. For more detailed analyses, appropriate corrections will need to be found to compensate these fluctuations.
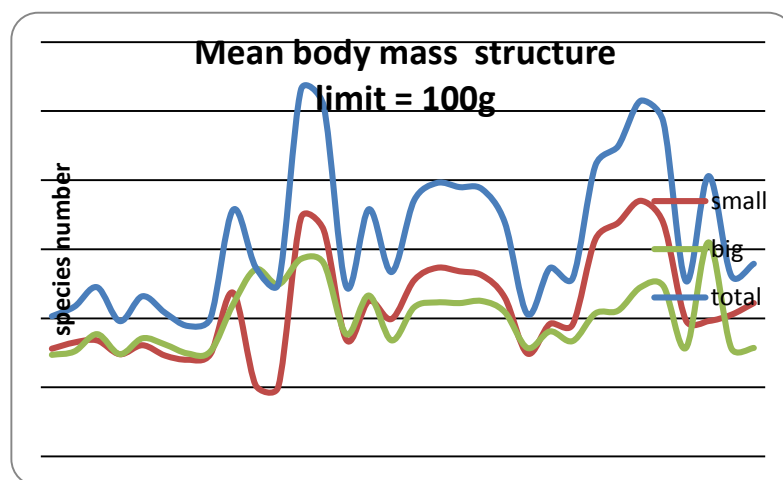
Figure 3.16. Total species richness and richness of species small and larger than 100 g found in the Portuguese Autum Bottom-trawl Survey; data courtesy to Hugo Mendes, IPMA. Flucutations are likely to be largely reflections of variations in methodology.

For the Barents Sea, we obtained data covering intensive surveys in the years 2006 to 2012, thus substantially enhancing the evidence base compared to our analysis from 2013, which was relying on data from the year 2009 only. The cumulative species-size distribution over these surveys clearly follows a Pareto law (Figure 3.17). Surprisingly, there is very little indication of a truncation of this distribution at large body sizes, suggesting that diversity in the Barents Sea has largely remained intact to the present day. The largest species included in the sample is the blue whale *Balaenoptera musculus.* As for the Portuguese trawl, however, sampling effort turned out to be highly uneven from year to year, and interanual comparison will require methods compensating for this.
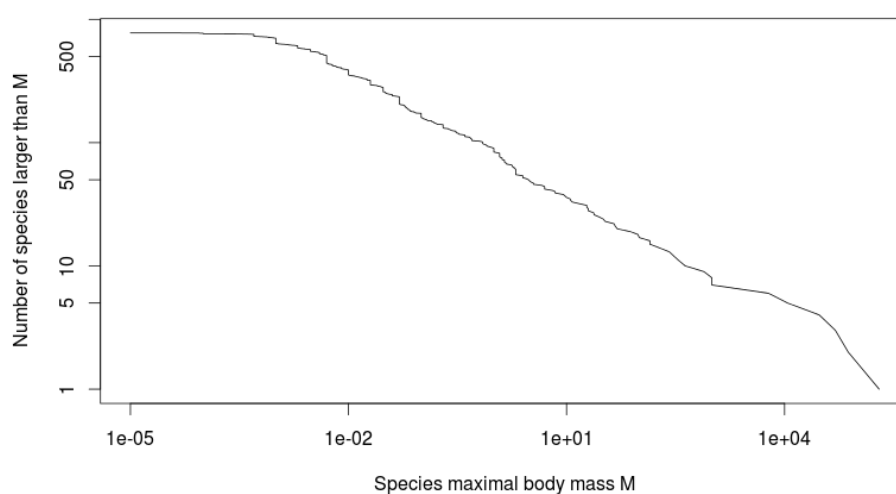
Figure 3.17. Pareto distribution of the body sizes (body mass in kg) of species over seven orders of magnitude, based on samples taken by the Norwegian research vessels at the Norwegian-Russian ecosystem surveys in the Barents Sea from 2006 to 2012 (Anon, 2009; 2010; 2011; Eriksen, 2012; courtesy to Lis Lindahl Jørgensen, IMR).

3.4.5 Indicators for the status of the benthos

An ecosystem component for which indicators are still insufficiently developed in the foodweb context is the benthic community. Of particular importance would be an indicator for the food availability to benthivorous fish. We note that some of the work proposed in Section 6 might lead to indicators that fill this gap.

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4 ToR b) Continue work on the large fish indicator, especially in waters other than the North Sea

This section of the report summarizes recent work to develop new Large Fish Indicators (LFIs) for marine regions where this approach has not previously been applied. Issues concerning the spatial scale of assessments based on the LFI are also addressed. The LFI was originally developed by ICES to support an Ecological Quality Objective (EcoQO) for the North Sea as part of OSPAR's pilot project to establish a framework to support an ecosystem approach to management. The intention was subsequently to role the process developed in the North Sea out to the other OSPAR regions. The LFI is considered only within its original EcoQO context. Consideration of the LFI as a potential foodweb indicator to support implementation of the MSFD is covered in Section 3 of the report addressing ToR a.

4.1 LFI method and application

The LFI method is generally well described in the literature, but unfortunately, the common practice of not weighing all species by length group leads to an issue with estimating weight-at-length for all species. The working group suggested that if weight-at-length is not recorded at the survey, the following methods could be used to estimate weight at length in order of relevance:

1) Species-specific length–weight relationships obtained from the surveyed area in other years;
2) Species-specific length–weight relationships obtained from the larger region;
3) Species-specific weight–length relationships from other regions;
4) Length–weight relationships of similar shaped species in the region;
5) Weight=0.01*length^3.

It should be considered that moving down the list makes the results less accurate, and that the proportion of biomass which is estimated using option 5 should be small (<5–10%).

4.2 Overview of LFIs

The following section presents an overview of the development of LFI indicators focusing on areas other than the North Sea and Celtic Sea, where results have already been published. It is possible that more work is ongoing, and WGECO encourages that this work is presented for inclusion in an updated overview.

4.2.1 Southern Bay of Biscay

Last year, WGECO reported on the early stages of development of an LFI in for the southern Bay of Biscay region (WGECO, 2013). We now report on the final outcome for this LFI (Modica *et al.*, in press).

The Southern Bay of Biscay LFI is derived using data collected by the DEMERSALES experimental bottom-trawl survey (ICES code: SPNGFS) carried out annually as part of the ICES IBTS Northeastern Atlantic area (ICES Areas VIIIc and IXa) by the Spanish Institute of Oceanography in the southern Bay of Biscay (Figure 4.1). Standardized

data were available from 1990 and data up to 2010 were analysed. Between 104 and 117 trawl samples were obtained between September and October each year from a depth range of 70 m to 500 m following depth-stratified random sampling design. Following previously established protocols (Greenstreet *et al.*, 2011; Shephard *et al.*, 2011), the suite of species included in the LFI was selected and a length threshold defining "large fish" of 35 cm was established. Figure 4.2 shows the temporal trend in the resulting LFI.
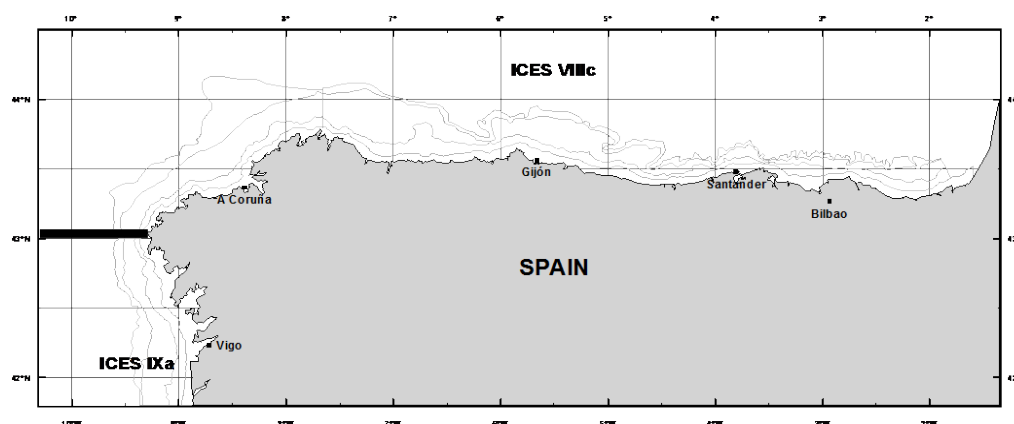
Figure 4.1. Study area covering ICES Area VIIIc in the southern Bay of Biscay region off the northwest coast of Spain.
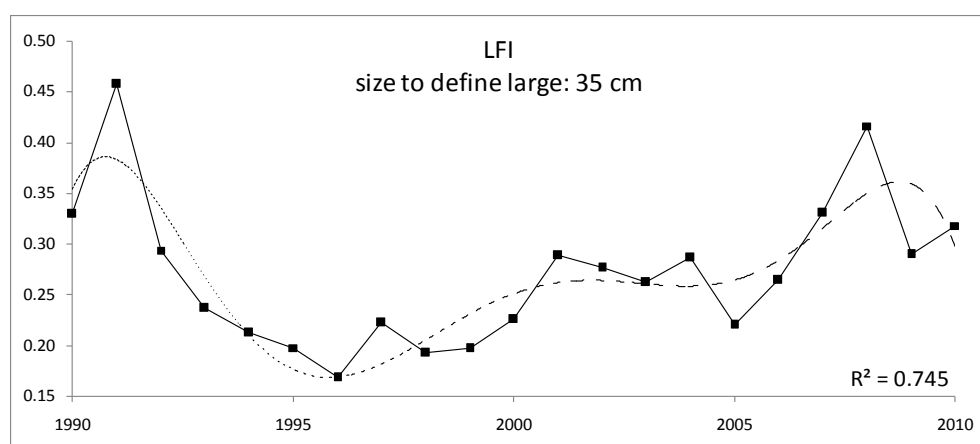
Figure 4.2. Variation in the LFI for the southern Bay of Biscay area between 1990 and 2010. Dotted line shows the 6th degree polynomial smoother fitted to the time-series based on a "large" fish defining threshold length of 35 cm, giving a fit of r^2=0.745. LFI trends based on "large" fish defining threshold lengths of 30 cm, 40 cm and 45 cm showed similar trends, but the 6th degree polynomial fits, at r^2=0.699, r^2=0.704 and r^2=0.690 respectively, were weaker, suggesting that 35 cm was the most appropriate "large" fish defining threshold length for the fish community in this marine region.

Following the procedures established by Greenstreet *et al.* (2011) and Shephard *et al.* (2011), annual indicators of community averaged of fishing mortality ($\overline{F}_{com,Y}$) and spawning–stock biomass ($\overline{B}_{com,Y}$) were derived for the Southern Bay of Biscay based on data for four assessed commercial stocks in the region (white anglerfish, black-bellied anglerfish, horse mackerel, and hake):

$$\overline{F}_{com,Y} = \frac{\sum_{s=1}^{4} F_{s,Y} / F_{s,msy}}{4} \quad \text{and} \quad \overline{B}_{com,Y} = \frac{\sum_{s=1}^{4} B_{s,Y} / B_{s,msy}}{4}$$

$F_{s,Y}$ and $F_{s,msy}$, are respectively the annual estimates of fishing mortality for each species in each year and the MSY reference values for fishing mortality for each species, and $B_{s,Y}$ and $B_{s,msy}$, are respectively the annual estimates of spawning–stock biomass (SSB) for each species in each year and the MSY reference values for SSB for each species (ICES, 2013). In this instance, however, to bring the assessment more in line with the MSFD ethos, maximum sustainable yield (MSY) reference values were used rather than the precautionary (B_{pa} and F_{pa}) fisheries management reference values originally used by Greenstreet *et al.* (2011).

The LFI was designed to be as sensitive as possible to the impacts of fishing disturbance on demersal fish communities. Figure 4.3 shows the temporal trends in both the LFI and $\overline{F}_{com,Y}$, the indicator of fishing disturbance on the southern Bay of Biscay demersal fish community. It is clear that the two indicators do not immediately covary similar to results in both the North Sea and the Celtic Sea (Greenstreet *et al.*, 2011; Shephard *et al.*, 2011). The response of the Southern Bay of Biscay LFI to changes in community average fishing mortality was lagged by between five and eight years (Figure 4.4). Data were not available to explore lags longer than eight years.
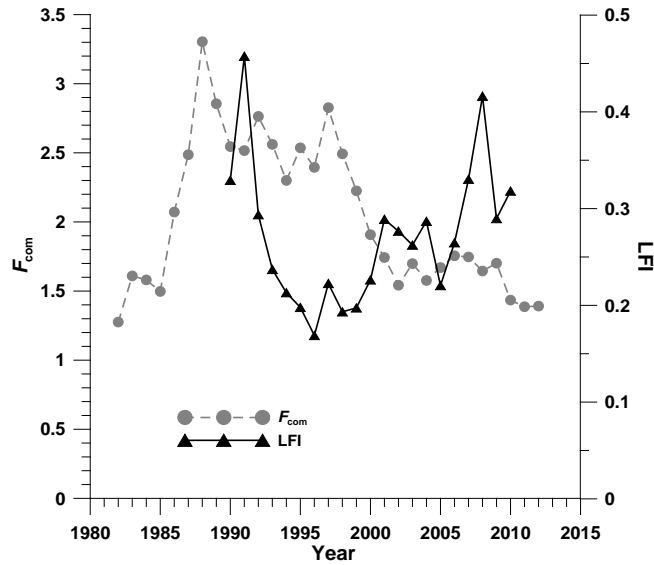
Figure 4.3. Plot showing temporal covariation in the community averaged fishing mortality index ($F_{com,y}$) and the LFI time-series.
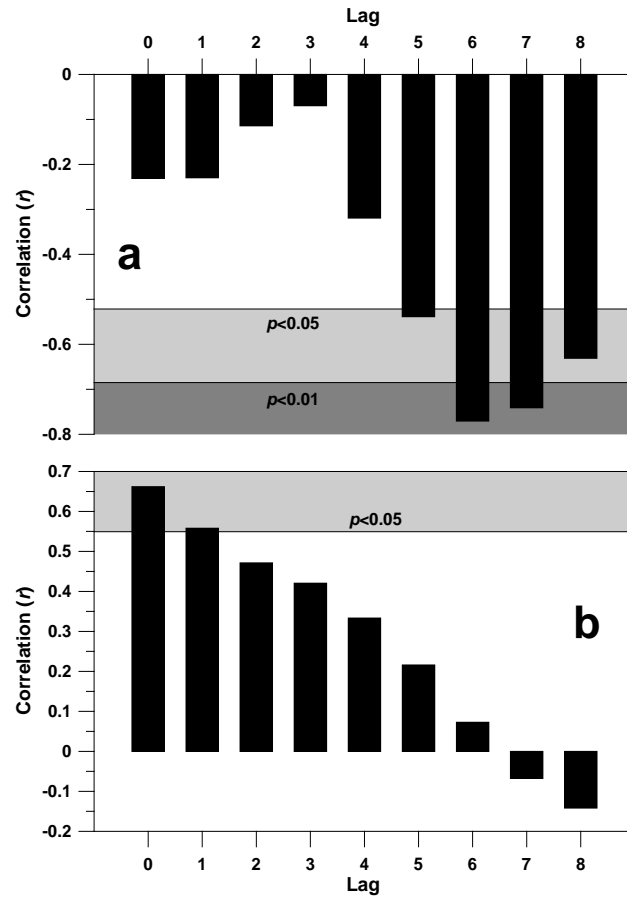
Figure 4.4. Variation in the correlation (expressed as Pearson's correlation coefficient *r*) between the LFI time-series and (a) the community averaged fishing mortality index, $F_{com,y}$ and (b) the community averaged spawning–stock biomass index, $B_{com,y}$ at various lags (e.g. LFI_y related to $F_{com,y}$ represents a lag of 0y, LFI_y related to $F_{com,y-1}$ represents a lag of 1y, LFI_y related to $F_{com,y-2}$ represents a lag of 2y, etc.). The 5% and 1% significance levels are shown based on a one-way tests with 9 degrees of freedom for $F_{com,y}$ and 8 degrees of freedom for $B_{com,y}$, assuming effective sample sizes of 11 and 10 respectively derived from application of the Chelton method to account for time-series autocorrelation.

The best-fitting lagged relationships (6 y lag with $\overline{F}_{com,Y}$ and 0 y lag with $\overline{B}_{com,Y}$) were used to determine management targets for the LFI that would be equivalent to fisheries management meeting targets stipulated for the four assessed stocks under the MSFD ($\overline{F}_{com,Y} < F_{s,msy}$ and $\overline{B}_{com,Y} > B_{s,msy}$, or $\overline{B}_{com,Y} > 0.5B_{s,msy}$) (Figure 4.5). While the MSFD might require stocks to be at B_{msy} to achieve GES, fisheries managers recognize that setting targets for SSB is impractical because stock size is influenced by factors outside their control, such as environmental influence on recruitment. Instead they set limits for SSB, such that should SSB fall below a "trigger" level ($B_{trigger}$), immediate management action should be implemented to halt further decline, such as severe limitations on fisheries responsible for the decline. An alternative target for the LFI, equivalent to maintaining $\overline{B}_{com,Y} > B_{trigger}$, where $B_{trigger} = 0.5B_{s,msy}$ is assumed. These relationships suggest that an LFI target equivalent to fishing at MSY might lie between 0.35 and 0.41, almost identical with LFI targets of between 0.34 and 0.41 equivalent to maintaining stocks above $B_{trigger}$. Should the four stocks reach B_{msy}, the LFI could rise to a value over 0.60.
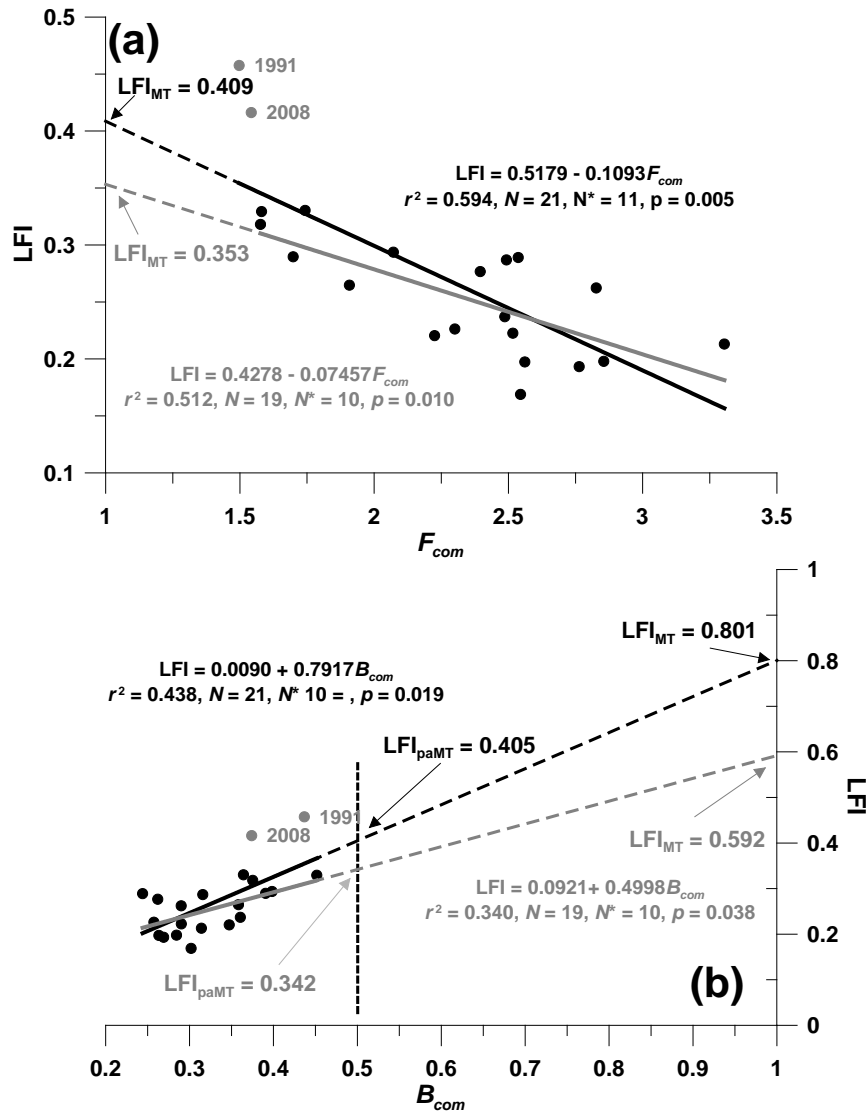
Figure 5. (a) Six-year lagged relationship between the LFI and the community averaged fishing mortality indicator (F_{com}) and (b) relationship between the LFI and the community averaged stock biomass indicator (B_{com}). Two linear regression relationships are shown in both panels, one for all data points (black line) and one excluding the 1991 and 2008 outlier data points (grey). Dashed lines show extrapolation of the linear regression relationships to estimate potential management targets (LFI$_{MT}$) at $F_{com} = 1.0$ and $B_{com} = 1.0$. In panel (b), potential precautionary management targets (LFI$_{paMT}$) at $B_{com} = 0.5$ (equivalent to $B_{trigger}$) are also indicated.

Once more adopting the approach outlined by Greenstreet *et al*. (2011) and Shephard *et al*. (2011), the lagged relationships with $\overline{F}_{com,Y}$ were combined to derive a statistical forecast model to predict future change over the eight years following the end of the current fisheries mortality *time-series*, based on changes in $\overline{F}_{com,Y}$ that occurred between 2005 and 2012 (Figure 6). This model suggests that a target of 0.35 for the southern Bay of Biscay LFI might be achieved in 2017. Recent reductions in fishing mortality have almost been sufficient to meet the LFI target in the near future; only further relatively minor reductions in fishing mortality might be necessary to ensure that the LFI is maintained above the target.
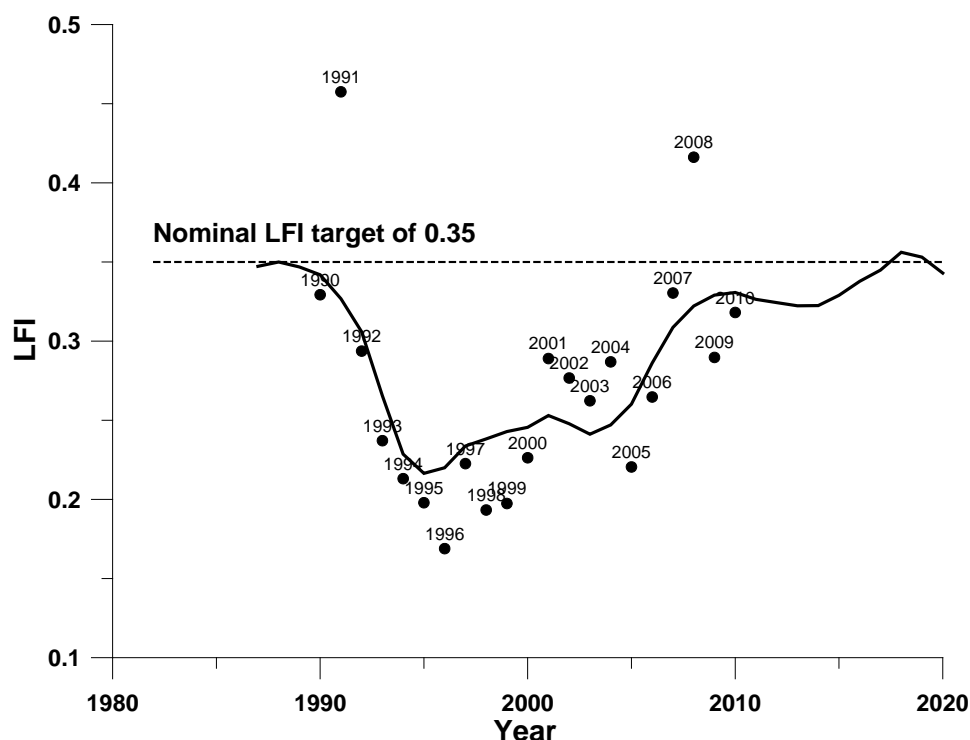
Figure 6. Modelled trend in the LFI based on the average outcome of the four significant lagged relationships, for five-, six-, seven- and eight-year lags, with F_{com}. Solid line shows the modelled trend and dots show actual observed annual values. A nominal LFI target of 0.35 is indicated by the dashed line.

4.2.2 Central–southern Tyrrhenian Sea (geographical Subarea 10)

WGECO received the materials for this section from Maria-Teresa Spedicato and Isabelle Bitetto, COISPA, Bari Italy.

The objective of this exercise was firstly to undertake an exploratory analysis to determine the appropriate threshold length that defines "large fish" in order to derive the LFI in Mediterranean areas, and secondly, to compare differences in the performance of the LFI when basing the indicator on abundance (number of individuals per square km) or biomass (weight of fish per square km), the latter being the metric suggested in the Commission Decision (6 November 2008). In the Central-Southern Tyrrhenian Sea, the LFI was initially derived using numbers instead of weight of fish, since early on in the MEDITS survey time-series, individual fish weight was not required as part of the survey standard protocol, but has been introduced in 2012.

The central-southern Tyrrhenian Sea (Geographical Subarea 10) was selected as a case study and the MEDITS groundfish survey time-series from 1995 to 2013 was used to derive the LFI. Fish and elasmobranchs that were sampled since the beginning of the time-series were retained in the list of the selected species on which the LFI was based (Table 4.1), while crustaceans and cephalopods were excluded. In a second explorative analysis, the bentho-pelagic species, *Trachurus trachurus*, *Trachurus mediterraneus* and *Micromesistius potassou*, were excluded. To avoid spurious influence of rare species, those species that occurred in less than 5% of the samples were excluded from the analysis. Three thresholds defining "large fish" (L_{LFTL}) were explored, 20 cm, 30 cm and 40 cm, and the LFI ($I_{LF,y,N}$ or $I_{LF,y,B}$) was calculated both on the basis of number (*N*) and weight (*B*) of individual fish as

$$I_{LF,y,N} = \frac{N_{>LFTL,y}}{N_{>LFTL,y} + N_{\leq LFTL,y}} \quad \text{and} \quad I_{LF,y,B} = \frac{B_{>LFTL,y}}{B_{>LFTL,y} + B_{\leq LFTL,y}}$$

where $N_{>LFTL,y}$ (or $B_{>LFTL,y}$) is the number (or biomass) of fish sampled in each year greater than the chosen large fish threshold length and $N_{\leq LFTL,y}$ (or $B_{\leq LFTL,y}$) is the number (or biomass) of fish sampled in each year less than or equal to the chosen large fish threshold length.

Table 4.1. List of species included in the LFI suite.

Citharus linguatula	*Pagellus bogaraveo*
Helicolenus dactylopterus	*Pagellus erythinus*
Lepidorhombus bosci	*Phycis blennoides*
Lophius budegassa	*Raja clavata*
Lophius piscatorius	*Solea vulgaris*
Merluccius merluccius	*Spicara flexuosa*
Micromesistius potassou	*Trachurus mediterraneus*
Mullus barbatus	*Trachurus trachurus*
Mullus surmuletus	*Trisopterus m. capelanus*
Pagellus acarne	*Zeus faber*

Using a large fish threshold length (LFTL) of 20 cm produced an LFI with the lowest coefficient of variation (CV), but the LFI seemed more influenced by possible peaks in the occurrence/weight of some species. Similar patterns, both when based on numbers and weight were observed for LFI trends when LFTLs of both >30 cm and >40 cm were used (Figure 4.7). The influence of bentho-pelagic species was negligible regardless of whether the LFI was determined based on numbers or weight (Figure 4.8).
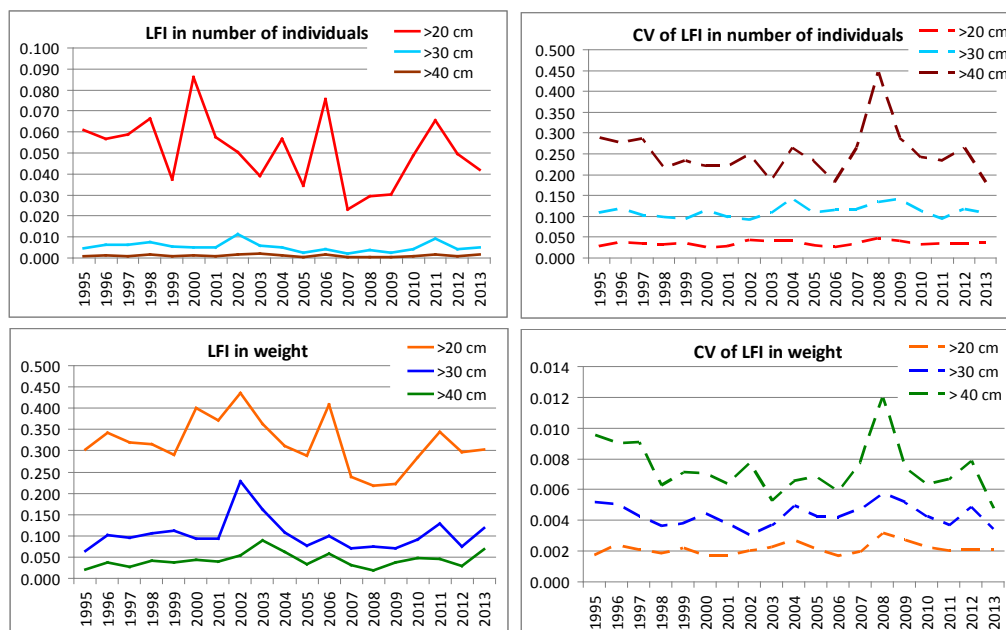
Figure 4.7. LFI in number of individuals and weight (left panels) and relative coefficient of variations (right panels) for three thresholds: 20, 30 and 40 cm.
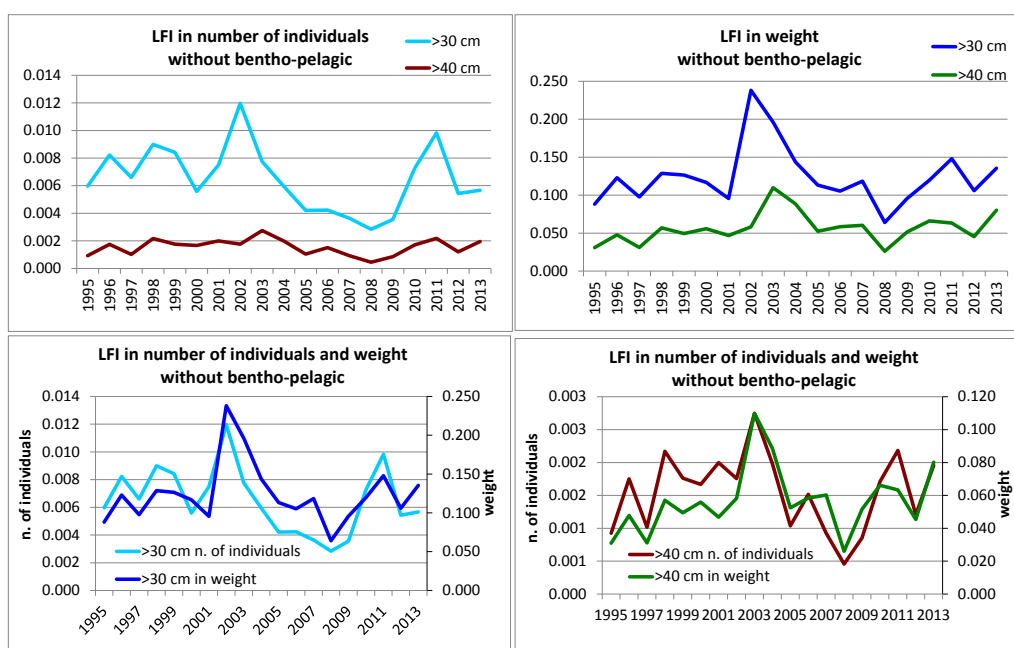
Figure 4.8. LFI trends based on numbers and weight of fish for the two thresholds, >30 cm and >40 cm, without bentho-pelagic species.

In Figure 4.9 the relative contributions of the 20 sampled species is represented along with the respective LFI trends based on LFTLs of both >30 cm and >40 cm. Using a LFTL of >30 cm and based on numbers, *Merluccius merluccius*, *Phycis blennoides*, *Lophius budegassa* and *Raja clavata* are the key species influencing the LFI. *Pagellus erythinus* is important in two years, when a spawning aggregation was sampled. Similar results were obtained when a LFTL of >40 cm was used.
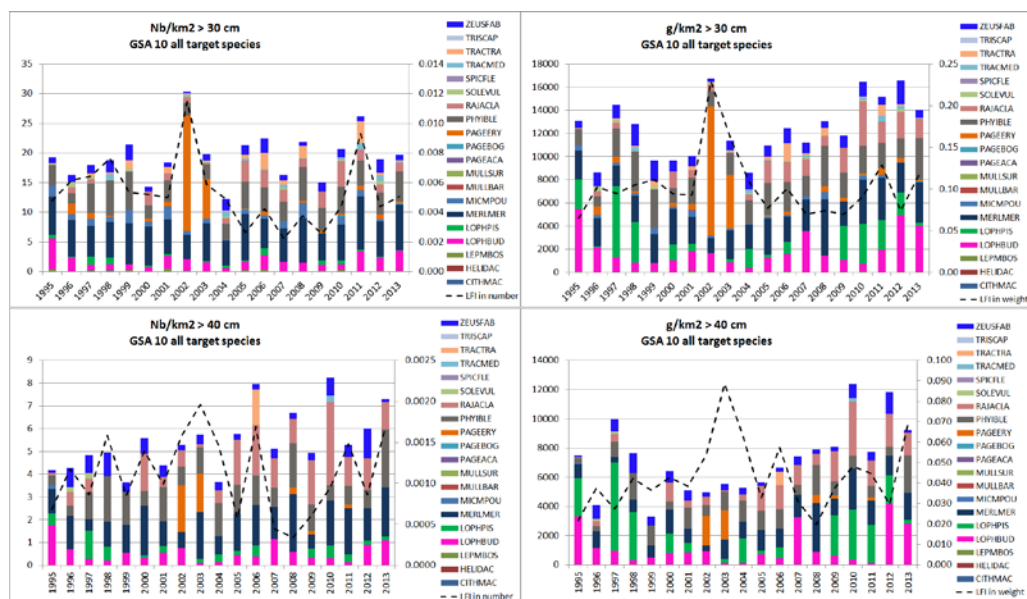
Figure 4.9. LFI trends based on numbers and weight of individuals for large fish length thresholds of >30 cm and >40 cm including all main sampled fish species) and the total (all lengths) relative contribution of each of each species in each year.

From this preliminary analysis, an LFTL of 30 cm seems most suitable for both the metrics, based on weight and number of individuals. The LFI is principally influenced by a relatively small number of species (about four), but is still a useful indicator for the broader "fish community". Further analysis in other Mediterranean geographical subareas is required to make the indicator applicable across a wider geographic range.

4.2.3 The Baltic Sea

The development of the LFI for the western Baltic Sea (SD 22–24) (Figure 4.10) based on the Baltic International Trawl Survey (BITS) has been reported previously (WGECO 2012; 2013). A cut-off length of 30 cm was chosen for this LFI resulting in a significant negative correlation to fishing mortality with a lag of two years. The LFI of the demersal fish community, represented by eight fish species, was highly dominated by cod making up 67% of the total biomass and 84% of the biomass of fish >30 cm TL. When cod was removed from the LFI analysis the same pattern remained, suggesting that the cod fishery exerted a similar size selection on the other fish species. The LFI shows an increasing trend over the last years, but may be expected to decline in the coming years due to the recent severe decrease in growth rate of cod (WGBFAS 2014). No target has been defined.
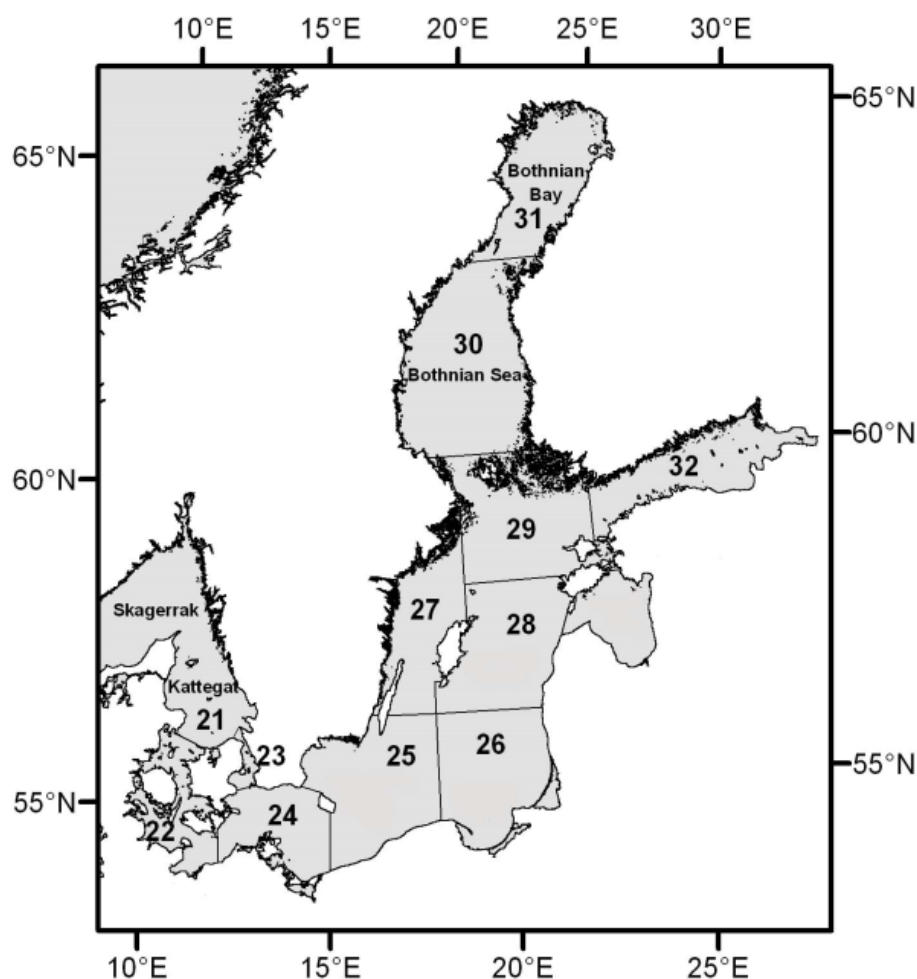
Figure 4.10. Map of the Baltic Sea showing the subdivisions referred to in the text. The Baltic International Trawl Survey (BITS) operates in SD 22–29.

There is also ongoing work developing an LFI for the Polish EEZ of SD 25 and 26 in the eastern Baltic Sea as part of developing an LFI for Poland's MSFD assessment. In this work, separate GES-boundaries have been proposed for the two subdivisions based on expert judgement. According to the preliminary assessment, the LFI has been increasing from 2001 to 2011, and GES was reached during the last years (2009–2011) in both areas. Progress in these LFI assessments will be reported through a live core indicator report on the HELCOM website (Oesterwind *et al*., 2013).

WKIND (2013) calculated DCF-indicators for ecosystem effects of fisheries where data were available, including the $LFI_{>30\ cm}$ for the Baltic Sea based on all available BITS data (SD 22–29 excluding 23) from 2001 and onwards. The length threshold and species list from the western Baltic LFI calculations were used for the calculations. This preliminary analysis showed a steady increase in the LFI from 2001 to 2012 (Figure 4.11). There was a short lagged cross-correlation between F_{com} (i.e. F for eastern and western Baltic cod) and the LFI (Figure 4.11), however, the lag at 0 yrs is dubious as the impact of F_{com} on the size structure of a fish community should be lagged by at least one year (Greenstreet *et al*., 2011). The pre-whitened time-series (method see Probst *et al*., 2012) did not indicate any significant correlations (Figure 4.11). Due to the late standardization of the survey gear (since 2001) the analysis covers only eleven years. With such a short time-series it is difficult to identify significant pressure

state relationships. An LFI indicator including the whole BITS survey area of the Baltic Sea will require more testing before being operational, including the definition of a target.
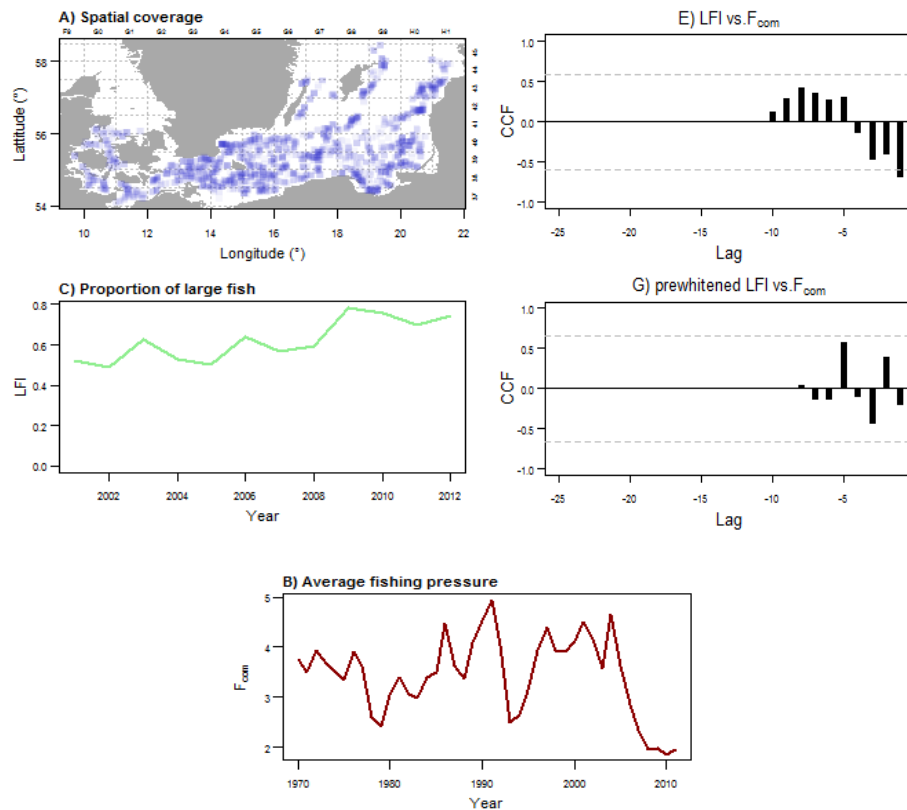
Figure 4.11. Spatial coverage of the BITS sampling used to calculate LFI$_{>30\ cm}$ for the Baltic Sea. F$_{com}$ is the combined fishing mortality for the eastern and western Baltic cod. Cross-correlation between F$_{com}$ and the LFI with and without correction for auto correlation.

4.2.4   The eastern North Sea

For the purpose of the MSFD evaluation, some countries are applying assessment areas smaller than the subregional scale. Sweden has proposed to use the Skagerrak (ICES Subdivision 20, SD20), the Kattegat (SD21) and the Sound (SD23) as assessment areas of the Greater North Sea (Figure 4.12). In this case, the rationale for also exploring the division of an LFI assessment into smaller geographical subareas is both ecological and managerial. ICES Area IIIa including the sound is characterized by decreasing salinity and depth towards the Baltic with associated changes in fish assemblage structure. Some of the important commercial species such as cod and plaice are divided into North Sea stocks including the Skagerrak and into Kattegat stocks with the potential for fisheries management to operate at this geographical scale. The sound is a geographically well-defined area where technical regulations have prevented the use of trawls since the 1930s, and there is also evidence of a productive cod subpopulation unit in the area (Sundelöf *et al.*, 2013).

Figure 4.12. Map showing ICES Division IIIa with the Skagerrak SD20, the Kattegat SD21, and the Sound SD23 with adjacent areas.

To investigate how the LFI has developed over time, samples from the International Bottom-trawl Survey (IBTS) in Division IIIa were split among the three subareas. Large fish were defined as being over 40 cm TL and the LFI was calculated as the proportion by weight of large fish in the sample following Greenstreet *et al.* (2011).

In the Sound, the LFI has remained well above the North Sea threshold (set at 0.3 following Greenstreet *et al.*, 2011) for the years sampled (Figure 4.13). The Skagerrak and the Kattegat exhibit similar historical declines of the LFI during the 1980s and 1990s. The LFI in the Skagerrak shows sign of a slight recovery over the last ten years, albeit not reaching the level of 0.3 (Figure 4.13), similar to the LFI for the North Sea as a whole (Greenstreet *et al.*, 2012). The LFI in the Kattegat remains at the lowest level in the time-series.
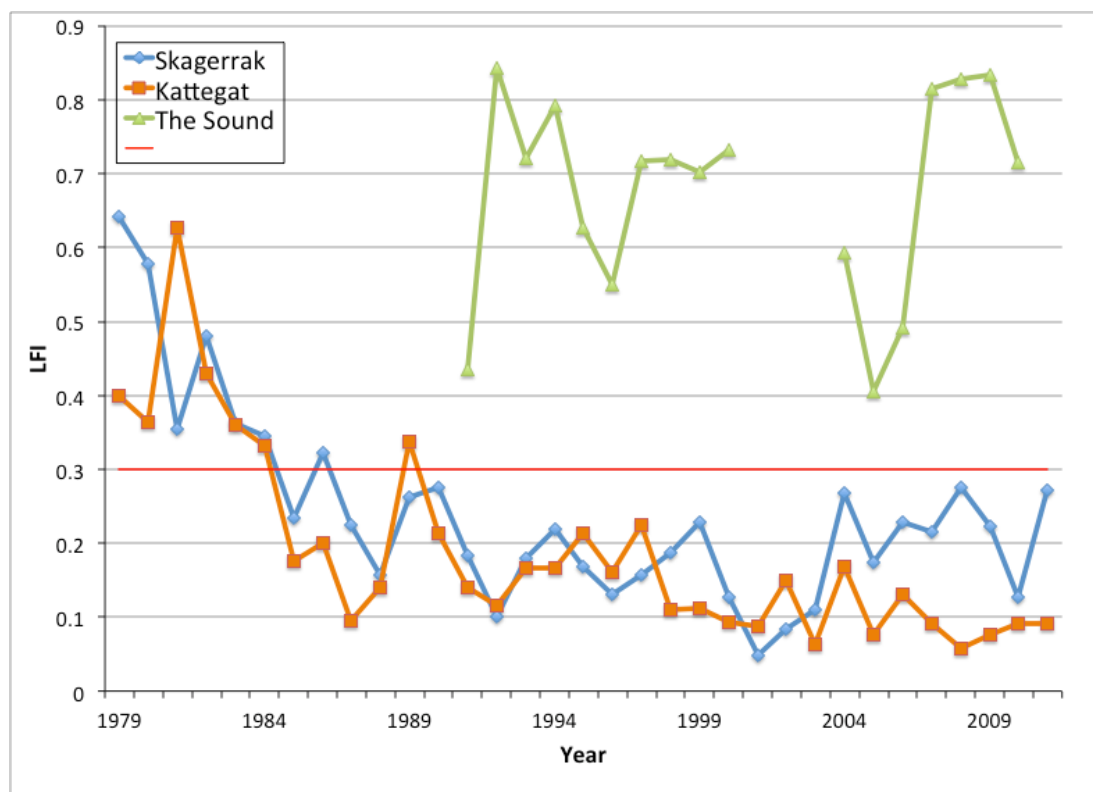
Figure 4.13. The large fish indicator (LFI) in three subareas of the North Sea: Skagerrak SD20, Kattegat SD21 and The Sound SD23.

The divergent trends for the different subareas are partly explained by the difference in state of the local cod stocks. The local subpopulation of cod in the sound is characterized by a wide age/size structure, indicating a moderate fishing pressure, compared to the adjacent stocks (Lindegren *et al.*, 2013). The North Sea cod stock, which is present in the Skagerrak is showing signs of recovery, whereas the Kattegat cod SSB has been at a historically lowest level since 2000 (ICES, 2013). Separating the assessment for the LFI into smaller subareas may thus provide additional information for management. With the present sampling scheme of the IBTS, a division of the assessment into more subareas will inevitably lead to reduced replication, which could make it more difficult to detect changes.

4.2.5   Overview of LFIs

An overview of the currently published LFIs and ongoing work known by WGECO is given in Table 4.2.

Table 4.2. Overview of currently published regional LFIs and ongoing work.

AREA	LFI DEVELOPMENT STAGE	TIME–SERIES	SPECIFIC TRESHOLD DEFINED	SPECIFIC REFERENCE LEVEL
North Sea	Completed[1]	Yes	Yes	Yes
Celtic Sea	Completed[2]	Yes	Yes	Yes
Southern Bay of Biscay	Completed[3]	Yes	Yes	Yes
Central-Southern Tyrrhenian Sea	Ongoing[4]	Yes	No	No
Baltic Sea	Ongoing[5]	Yes	Yes	No
Poland EEZ	Completed[6]	Yes	Yes	Yes
Kattegat North	Ongoing[7]	Yes	No	No
Kattegat South	Ongoing[7]	Yes	No	No
The Sound	Ongoing[7]	Yes	No	No
Gulf of Cádiz	Ongoing[8]	No	No	No

References: [1] **Greenstreet** *et al.*, **2011**, [2] **Shephard** *et al.*, **2011**, [3] **see Section 4.2.1**, [4] **see Section 4.2.2**, [5] **WGECO 2012; 2013**, [6] **Oesterwind** *et al.*, **2013**, [7] **see Section 4.2.4**, [8] **WGECO 2012**.

4.3   Regional scale LFI assessments

The Marine Strategy Framework Directive defines the spatial scale on which Good Environmental Status (GES) must be achieved. Member States sharing Regions must cooperate through the auspices of the Regional Seas organizations (OSPAR, HEL-COM) to achieve GES at these spatial scales. This presents challenges for assessment programmes for taxa, like fish, whose communities bridge the individual exclusive economic zones of different Member States. There are two options for these case; sub-region specific LFIs, each with a separate threshold size, or regional LFIs which are then consistent for all Member States. The question of subregion specific LFIs also occurs when an area is covered by two different surveys, each sampling only part of the total area. For example, the Northeast Atlantic Region includes the Celtic Seas and the Greater North Sea subregions. In the latter, fish community monitoring is facilitated by well-integrated international trawl surveys that cover much of the sub-region. In the former, national surveys are more closely linked to Member State's wa-ters; consequently different surveys covering different areas are undertaken in different seasons following different protocols. In order to assess status at the scale of the whole Celtic Seas subregion, some means of integrating the information provided by these disparate surveys is necessary. In these cases, it is relevant to know whether regional scale LFIs in general reflect subregional LFIs as well as whether LFIs are comparable between surveys covering different parts of the same region.

A recent study addressed this issue using data collected from three surveys operating in the North Sea at different times of the year using different trawl gears: the first quarter (Q1) International Bottom-trawl Survey (IBTS), third quarter (Q3) IBTS, and Q3 Dutch Beam Trawl Survey (DBTS) (Greenstreet *et al.*, in review). Since each survey

covered most of the Greater North Sea area (except the English Channel, and the far northern North Sea in respect of the Q3 DBTS), single regional-scale assessments of the status of North Sea demersal fish based on the Large Fish Indicator (LFI) could be supported by each survey (Figure 4.14). The Greater North Sea was split into five subdivisions (Figure 4.14) and separate LFI trends were derived for each spatial sub-division using each of the surveys (using both the North Sea standard large fish threshold length of 40cm and survey specific thresholds length determined for the two Q3 survey following established protocols (Greenstreet *et al*., 2011; Shephard *et al*., 2011)), thereby mimicking the situation where data could only be obtained from separate and variable subregional-scale data sources. Differences in the LFI trends derived using the different surveys for each spatial subdivision of the Greater North Sea were examined and various methods of integrating these separate spatial subdi-vision LFI trends were tried to determine which best replicated the single regional-scale assessment trends. Correlation analysis was used to compare LFI time-series and the Chelton procedure applied to take account of autocorrelation within each time-series and determine the effective sample size. Since non-significant correlations were of as much interest as significant correlations, a Benjamini and Hochberg correc-tion procedure for multiple comparisons was applied. While this procedure reduces the risk of type I errors, it also minimizes the risk of "false negatives".
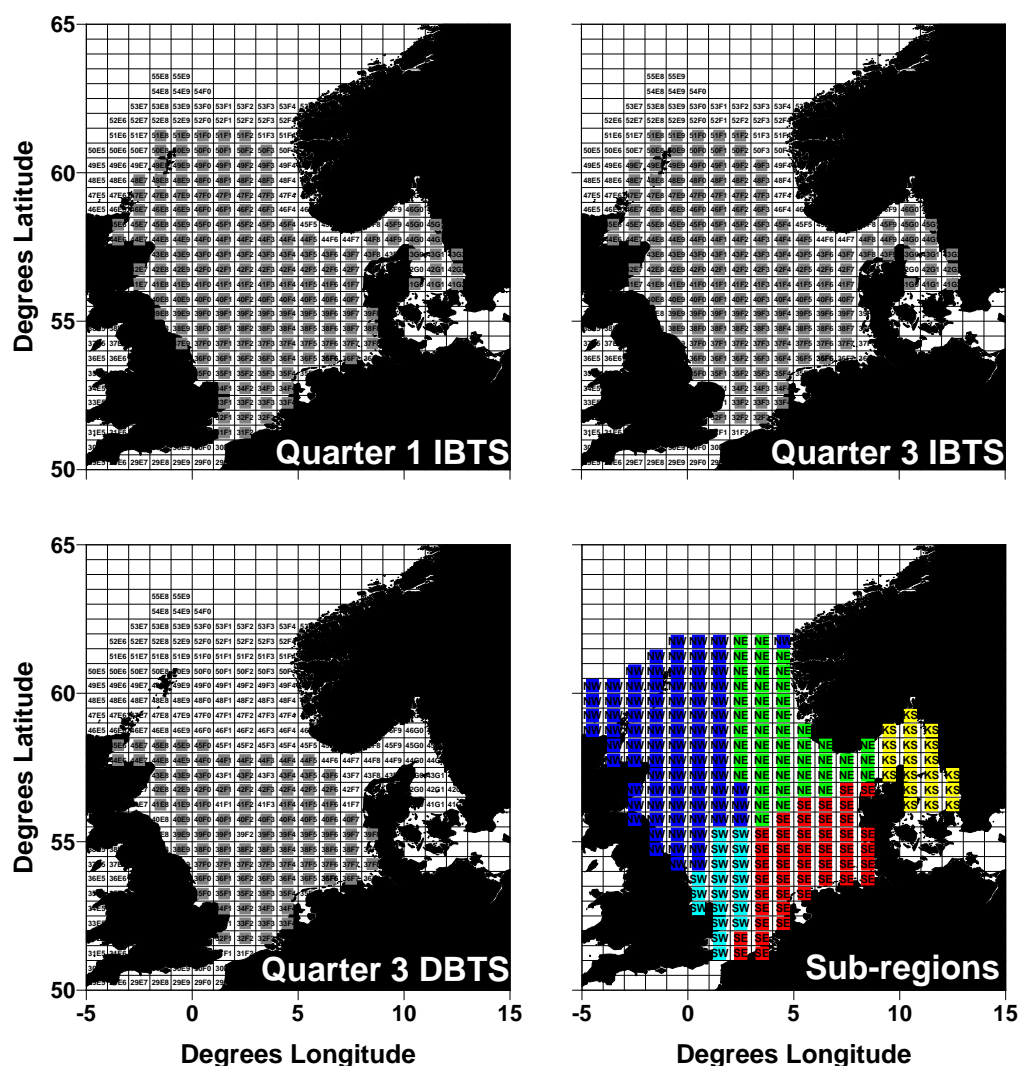
Figure 4.14. Charts showing the standardized sampling areas covered by the Quarter 1 and Quarter 3 International Bottom-trawl Surveys (IBTS), the Quarter 3 Dutch Beam Trawl Survey (DBTS), and the subdivision of the Greater North Sea into five subregions: the Kattegat-Skagerrak (SK), Northeast (NE), Northwest (NW), Southeast (SE) and Southwest (SW).

Marked variation was observed between the different subregions in individual subregional LFI trends derived from the three surveys, using both standard and survey-optimized LFTLs (Figure 4.15). Using Q3 IBTS or Q3 DBTS data, only one correlation between subregion pairs (northwest and southeast survey-optimized LFTL Q3 DBTS) was statistically significant (Table 3), and in this instance the relationship was negative. Although not significant, all four correlations comparing the northeastern and southeastern subregions, using both Q3 surveys and either LFTL, were negative. Given that we anticipated the majority of correlations to be positive, the number of negative correlations between northern and southern subregions suggests that demersal fish community LFIs in the northern and southern halves of the North Sea followed different temporal trajectories.
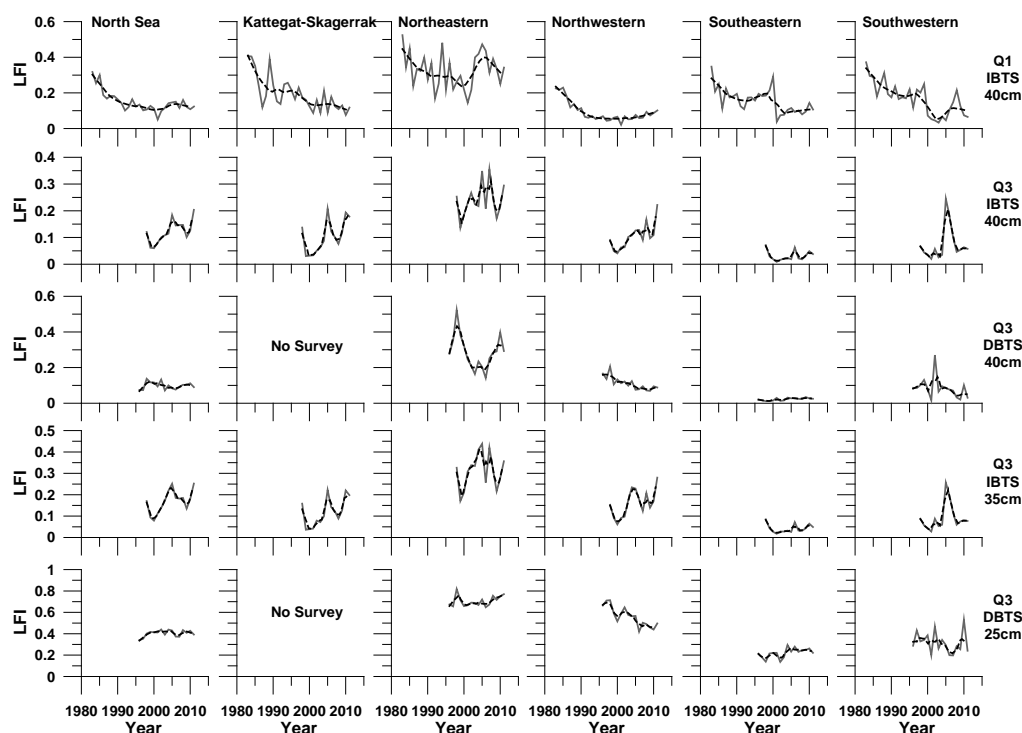
Figure 4.15. Trends in the regional scale LFI determined using three different surveys and the standard "large fish threshold length" (LFTL) of 40 cm and survey specific LFTLs for the Q3 IBTS and Q3 DBTS of 35 cm and 25 cm respectively, compared with equivalent trends derived for five separate subregions: Kattegat-Skagerrak, northeastern, northwestern, southeastern and southwestern North Sea. Grey line shows actual data while the black dashed line shows a Lowess smoother fitted to the main time-series data.

Significant positive correlations between pairs of subregional LFI trends were only observed when using Q1 IBTS data (Table 4.3). Enhanced statistical power associated with the longer Q1 IBTS time-series (larger sample size) helped but was not the full explanation. Starting earlier, the Q1 IBTS included the period of LFI decline. Most of the regional-scale decline had occurred by 1992 and, except for the northeastern subregion where little trend was evident, the pattern was similar in the subregions (Figure 4.15). The shorter Q3 DBTS and Q3 IBTS time-series commenced in 1996 and 1998 respectively, so missed this decline and only covered the period when the LFI was depressed and slightly increasing (Figure 4.15).

Correlations between paired subregion LFI trends based on Q1 IBTS data were more easily interpretable (Table 4.3). All correlation coefficients were positive. None of the correlations involving the northeastern North Sea were significant, marking this subregion as having a relatively unique LFI trend (Figure 4.15). The northwestern subregion LFI trend was also not correlated with either of the two southern North Sea subregional trends, corroborating the earlier observation that LFIs in the northern and southern halves of the North Sea followed different trajectories; a conclusion further underlined by the significant correlation between LFI trends in the two southern subregions. Not only was there little similarity between LFI trends for subregions located in different halves of the North Sea, but LFI trends in the two northern North Sea subregional were also quite dissimilar (Figure 4.15).

Despite differences between the subregional Q1 IBTS LFI trends, all five correlated significantly with the regional trend (Table 4.4), but the southeast, southwest and northeast subregion correlations were weak ($r^2 < 0.5$) and only significant because of

the statistical power provided by the longer Q1 IBTS time-series. Some comparisons involving the shorter Q3 IBTS and Q3 DBTS datasets actually generated large correlation coefficients, yet were not significant because of their smaller sample size. Conversely, correlation between the North Sea regional and northwestern North Sea subregional LFI trends was exceptionally close ($r^2 = 0.83$) (Table 4.4). A similar pattern emerged when basing the LFI on Q3 IBTS data, regardless of the LFTL used. The North Sea regional LFI trend did not correlate with the two southern North Sea subregional trends, but did correlate with the two northern North Sea subregional trends (Table 4). When using the survey-optimized LFTL of 35 cm, the correlation between the North Sea regional LFI trend and the northwestern North Sea subregional trend was again exceptionally close ($r^2 = 0.87$) (Table 4.4). Hence, regional scale LFI trends using either Q1 IBTS or Q3 IBTS data were only representative of trends in the two northern North Sea and the Kattegat-Skagerrak subregions. The reason could be that the regional scale LFI trend was primarily influenced by changes taking place in the northwestern North Sea subregion. The situation was even worse when using Q3 DBTS data; none of the correlations were significant. The Q3 DBTS regional LFI trend did not reflect changes taking place in any one subregion of the Greater North Sea.

Table 4.3. Correlation (*r*) coefficients for subregion pair LFI trend comparisons where the LFI is derived using three different groundfish surveys and using either the standard "large fish length threshold (LFTL) or the survey-optimized LFTL. The number of year's data in each paired comparison is given (*N*). The significance of each correlation coefficient was assessed following application of the Chelton procedure to determine the effective sample size (*N**), giving *N**-2 degrees of freedom.

SURVEY/LFTL	SUBREGION COMPARISON	*R*	*N*	SIGNIFICANCE
Q1 IBTS	KS v NE	0.340	29	Not Sig.
40 cm	KS v NW	0.671	29	$p = 0.008$ ($N^* = 14$)
	KS v SE	0.575	29	$p = 0.008$ ($N^* = 20$)
	KS v SW	0.640	29	$p = 0.008$ ($N^* = 16$)
	NE v NW	0.358	29	Not Sig.
	NE v SE	0.234	29	Not Sig.
	NE v SW	0.210	29	Not Sig.
	NW v SE	0.531	29	$p = 0.029$ ($N^* = 17$)
	NW v SW	0.631	29	$p = 0.021$ ($N^* = 13$)
	SE v SW	0.626	29	$p = 0.004$ ($N^* = 19$)
Q3 IBTS	KS v NE	0.436	13	Not Sig.
40 cm	KS v NW	0.570	13	Not Sig.
	KS v SE	-0.115	19	Not Sig.
	KS v SW	-0.077	19	Not Sig.
	NE v NW	0.408	14	Not Sig.
	NE v SE	-0.145	14	Not Sig.
	NE v SW	0.500	14	Not Sig.
	NW v SE	0.215	14	Not Sig.
	NW v SW	0.236	14	Not Sig.
	SE v SW	0.317	20	Not Sig.
Q3 IBTS	KS v NE	0.329	13	Not Sig.
35 cm	KS v NW	0.605	13	Not Sig.
	KS v SE	-0.141	19	Not Sig.

Survey/LFTL	Subregion comparison	R	N	Significance
	KS v SW	-0.088	19	Not Sig.
	NE v NW	0.556	14	Not Sig.
	NE v SE	-0.243	14	Not Sig.
	NE v SW	0.378	14	Not Sig.
	NW v SE	0.182	14	Not Sig.
	NW v SW	0.353	14	Not Sig.
	SE v SW	0.431	20	Not Sig.
Q3 DBTS 40 cm	KS			
	NE v NW	0.570	16	Not Sig.
	NE v SE	-0.404	16	Not Sig.
	NE v SW	0.022	16	Not Sig.
	NW v SE	-0.594	16	$p = 0.042$ ($N^* = 12$)
	NW v SW	0.302	16	Not Sig.
	SE v SW	-0.536	16	Not Sig.
Q3 DBTS 25 cm	KS			
	NE v NW	0.029	16	Not Sig.
	NE v SE	-0.238	16	Not Sig.
	NE v SW	0.119	16	Not Sig.
	NW v SE	-0.753	16	$p = 0.007$ ($N^* = 11$)
	NW v SW	0.247	16	Not Sig.
	SE v SW	-0.254	16	Not Sig.

Table 4.4. Correlation (*r*) parameters comparing the single regional scale LFI annual assessment trends with LFI assessments determined for each of five separate subregions: the Kattegat/Skagerrak region (KS), the northeastern North Sea (NE), the northwestern North Sea (NW), the southeastern North Sea (SE) and the southwestern North Sea (SW). LFI trends determined using Q1 IBTS, Q3 IBTS and Q3 DBTS data and various "large fish thresholds lengths" (LFTL) were examined. For each analysis the total sample size (*N*) is indicated along with the effective number of independent samples (*N**) determined using a modified Chelton procedure. Statistical significance is based on *N**-2 degrees of freedom.

SURVEY	LFTL (CM)	SUBREGION	R	N	N*	P
Q1 IBTS	40	KS	0.768	29	15	0.001
		NE	0.639	29	24	0.001
		NW	0.913	29	12	0.000
		SE	0.613	29	18	0.006
		SW	0.696	29	14	0.006
Q3 IBTS	40	KS	0.805	14	9	0.009
		NE	0.737	14	12	0.006
		NW	0.877	14	10	0.001
		SE	0.221	14	12	0.490
		SW	0.576	14	10	0.081
Q3 DBTS	40	KS				
		NE	0.579	16	15	0.024
		NW	0.333	16	15	0.225
		SE	-0.367	16	15	0.089
		SW	0.494	16	15	0.061
Q3 IBTS	35	KS	0.794	14	8	0.019
		NE	0.757	14	11	0.007
		NW	0.934	14	8	0.006
		SE	0.163	14	11	0.632
		SW	0.547	14	9	0.106
Q3 DBTS	25	KS				
		NE	0.318	16	15	0.248
		NW	-0.234	16	11	0.489
		SE	0.144	16	13	0.639
		SW	0.320	16	13	0.286

Single regional scale assessments appear inadequate as a means of monitoring change in the status of the demersal fish community across the whole Greater North Sea. The regional-scale LFI EcoQO might appear to be met, implying GES for North Sea demersal fish had been achieved when in reality the situation might only be acceptable in other parts of the North Sea. Conversely, the regional scale LFI might indicate sub-GES status regionally, when in reality several subregions could have reached GES. Performing subregional scale assessments would more accurately reflect actual circumstances, identify exactly where remedial management might be required, and perhaps identify particular management measures required; a reduction in beam trawl activity in the southern North Sea, or otter trawl activity in the northern North Sea, rather than a general reduction in fishing effort across the whole MSFD area.

Subregional time-series may differ for a variety of reasons, including the larger variation caused by the smaller number of samples taken in subregions (and hence potentially larger fluctuations due to random occurrences of one or two large catches of particular sizes) or difference in threshold lengths as well as underlying ecosystem differences. To ensure that the former has as small an effect and the later as large an effect as possible, WGECO recommends to base subregional analyses on ecologically relevant areas or habitats while considering that the areas should have sufficient samples.

4.4 References

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5 ToR c) Consider the ecosystem consequences of "balanced fishing" regimes

5.1 Context

The concept of "balanced fishing" has gained momentum over the recent years. The concern about the impacts of fishing on ecosystems and fisheries production has increased and lead to the development of ecosystem approaches to fisheries management. Remedies proposed to mitigate ecosystem impacts have included reducing fishing effort and, often, increasing selectivity. The latter was expected to reduce fishing impacts on non-target species, and to limit the waste of valuable natural resources. However, it has also been suggested that concentrating a given amount of fishing pressure on a restricted range of ecosystem components might generate more undesirable consequences than if this same amount would be spread more broadly across a diversity of components; each receiving a lower burden. There may be a "balanced fishing" regime across a range of species, stocks, and sizes that could mitigate adverse effects and address food security better than increased selectivity (Garcia *et al.*, 2012). To an extent the concept has been developed as an impulse to think of exploitation patterns at the community level and their potential ecosystem consequences, as opposed to managing stocks independently for maximizing economic or social benefits with ecosystem considerations as a constraint.

The approach has been one of the more hotly debated subjects in fisheries science for a few years, and a range of modelling papers produced results that suggest the concept works or possibly does not. The original paper (Garcia *et al.*, 2012) highlights a number of potential ecosystem benefits from balanced fishing, but does not apparently consider if there would be any non-beneficial impacts of the approach. The purpose of this ToR for WGECO was to consider whether all the likely impacts of a shift to balanced fishing would be potentially beneficial. In particular, whether partial progress towards balanced fishing would also be likely to deliver ecosystem benefits, as it is unlikely that perfectly balanced fishing could be achieved in practice.

This section sets out with a few definitions, since balanced fishing relies on several potentially ambiguous concepts. Considerations on how it could work are briefly raised. The core of the ToR is a review of model results and empirical evidence of ecosystem consequences of balanced fishing.

5.2 Definitions

Balanced fishing sounds smooth but may be understood in different ways. Garcia *et al.* (2012) proposed the following definition: "Balanced harvesting requires adjusting selectivity regulations to balance the impact of all fisheries in an area with the relative productivities of the species and sizes of fish in the ecosystem". This definition itself relies on several terms, such as selectivity and productivity, which are complex concepts with multiple dimensions and whose meaning might vary depending on the context. WGECO set out to first define these terms as they are used below, and to specify how the ToR question is understood.

5.2.1 Selectivity

Selectivity is defined as "the capacity of a fleet to capture a certain proportion of the fish population. This proportion is characterized by species, age and/or size".

Selectivity implies that the species and size composition of a fleet's catch differs from the composition of the population in a given area and time period. All fleets are species and size selective. Selectivity depends on a range of managerial and operational characteristics of the fleet, of which spatial, temporal and technological specifications are the main attributes. Selectivity may hence not exclusively be understood as the gear selectivity or even more narrowly as the mesh selectivity of codends in trawlnets. Nevertheless, most selection work relates to the catches of fish encountering the gear or more specific the netting of the gear. The described gear selection is hence mostly relative within the gear or relative across gears rather than relative to the actual present population, as this is difficult to estimate.

Recently the selectivity concept has been enriched by taking an ecosystem perspective and now sometimes also means the selective extraction by fishing activities of some ecosystem components from the environment. However, as suggested below, balanced fishing is not non-selective fishing; it relies on selective practices. To avoid ambiguities WGECO suggest that "selectivity" be used only in the restricted, technological sense and that other wordings be used when it comes to the distribution of fishing pressure across ecosystem components, such as "exploitation patterns" or "fishing regimes". The opposite of "balanced fishing" may not be "selective fishing" but for example "concentrated fishing".

5.2.2 Productivity

Balanced harvesting requires adjusting fishing pressure to the relative productivity of each species or size. To measure the degree of balance or to implement a balanced fishery would require a consistent measure of productivity across species and individuals.

Productivity is the rate of production of biomass per unit biomass per unit time, whatever the fate of this biomass; eaten or fished. A measure of productivity is the ratio of production to biomass over the lifespan of individuals, starting from a reference age (e.g. the age of recruitment). At steady state, production over the life cycle is equivalent to annual production summed over age classes. Somatic production (P) can be expressed as the product of mean numbers during the year $\overline{N}_a$ ($\overline{N}_a$) and change in body weight-at-age Δw_a summed over ages:

$$P_T = \sum_a \overline{N}_a \Delta w_a$$

The annual P/B ratio with no fishing, is a measure of productivity (neglecting reproduction); whereas P/B with fishing is the realized production. For size-structured populations, growth can be described with a growth-rate function, such that production is the product of mean numbers at weight and change in weight. This single-species context effectively assumes mean values of growth and mortality, ignoring that these rates may depend on interacting species.

In non-structured descriptions of species production, the P/B ratio is still a measure of productivity, but it needs to be calculated differently. In foodweb models the production of each consumer component (i) is the sum production of prey components (j) time the proportional allocation of a given prey to each of its consumers a_{ij}, times the transfer efficiency, te_i.

$$P_i = te_i \sum_j a_{i,j} P_j$$

Again, the P/B ratio is calculated by dividing production by the steady-state biomass of each trophic component.

A dynamic description for the change in biomass of a species with time is:

$$\frac{dB}{dt} = (\varepsilon f - r)B - mB$$

where ε is assimilation efficiency, f is the functional response, and r is the respiration rate. The first term on the right hand side of this equation represents production and $(\varepsilon f - r)$ is the P/B ratio.

5.2.3 Balanced fishing

Based on the considerations above, WGECO propose the following adjustments to the Balanced fishing definition:

Balanced harvesting requires adjusting exploitation patterns to balance the pressures of all fisheries in an area with the relative productivities of the species and sizes of fish in the ecosystem.

To WGECO, balanced harvesting applies to fisheries and their normal resources; fish and shellfish. Balanced harvesting is not generally meant to include other guilds such as plankton or top-predators; birds, dolphins or whales. In some places however, a definition limiting 'balanced fishing' to fish and shellfish would not cover the range of harvested resources. For example, Norwegian fisheries harvest a wider range of trophic levels, from plankton by a small pilot-fishery for *Calanus* in the Norwegian Sea up to whales and seals in the Barents Sea. In those cases "balanced fishing" disregarding the latter guilds may not pick up ecosystem responses to changed abundance of the target species.

Balanced fishing is balanced not just across size classes or trophic levels; it may also apply to species within a trophic level (Figure 5.1). If there would be a high level of compensation between species within a trophic level or functional group, then species might be exploited in an uneven way (Figure 5.1a). With low compensation within a functional group though, species dynamics would be independent or synchronized and the loss of the most exploited species would not be replaced by other species playing a similar role; then, a more balanced exploitation might be warranted (Figure 5.1b).
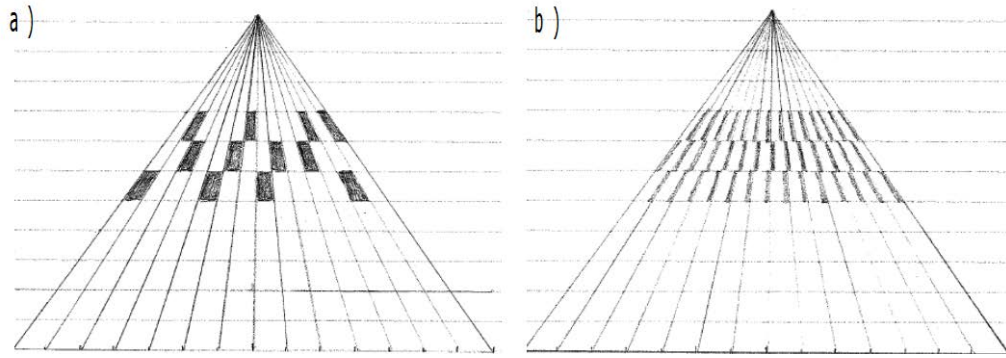
Figure 5.1. Balanced fishing across species. Fisheries exploit a few intermediate trophic levels in the trophic pyramid: the fish community. In a) a limited set of species are targeted and heavily exploited. Balanced fishing across fish species is illustrated in b): each species within the three fished trophic levels is exploited in a proportional way.

5.3   How could balanced harvesting work?

5.3.1   Some theoretical considerations

The balanced harvesting concept might not be easy to translate into practical fishing advice, as suggested by even simplistic modelling frameworks explored below.

5.3.1.1   The Lotka–Volterra model

The idea of balanced harvesting can, to a certain extent, be studied using simple Lotka-Volterra models of the form

$$\frac{dB_i}{dt} = \left(r_i - \sum_j G_{ij} B_j \right) B_i - F_i B_i ,$$

where B_i are time-dependent population biomasses, t represents time, F_i are exploitation rates and r_i and G_{ij} are parameters, and the indices i and j run over all *exploitable* species. It is not difficult to see that, if $B_i = B_i^*$ is a feasible (all $B_i^* > 0$) equilibrium solution of the unexploited ($F_i = 0$) system, then an exploitation pattern such that $F_i = c\, r_i$ for some constant $0 < c < 1$ leads to a new feasible equilibrium solution $B_i = (1-c)\, B_i^*$. If one considers the parameters r_i as some sort of production rates, such an exploitation pattern would therefore be seen as a form of balanced harvesting, and, indeed the biomasses of all exploitable populations would be reduced in proportion to each other. However, there are complications.

First, the Lotka-Volterra model above is an interaction model for the exploitable populations only, in which indirect interactions through common non-exploitable prey (benthos, zooplankton) are absorbed into the interaction coefficients G_{ij}. In principle, a re-arrangement of Lotka-Volterra models such as to treat some species explicitly and others implicitly (through "elimination of fast variables") is always possible, at least formally. However, depending on the set of exploitable species considered (all fish, all fish and shellfish, etc.), the values of all coefficients r_i and G_{ij} will generally differ. Thus, the coefficient r_i is not directly associated with a species; it has a meaning only in the context of a particular set of species. In particular, the r_i are not the productivities of the exploited species in the sense defined above. Instead, they com-

bine density-independent contributions from growth, reproduction, respiration, and mortality terms.

Second, there is no guarantee that when harvesting at rates $F_i = c\,r_i$, none of the populations that are treated implicitly will get extirpated. In the case of extirpations, their implicit treatment in the model breaks down (they would be assumed to have negative population sizes), and so would the balanced harvesting regime.

Third, there is no guarantee that all r_i will be positive, so that correspondingly the "balanced" exploitation rates $F_i = c\,r_i$ would be positive. In the classical two-species predator–prey version of the model, for example, r_i is negative for the predator. Since application of "negative exploitation rates" is impracticable, "balanced" harvesting in the sense that $F_i = c\,r_i$ will often be impracticable as well.

5.3.1.2 Other simple production considerations

A balanced fishery would harvest each species in proportion to its P/B ratio. In the age-structured case, yield-per-recruit would be a constant fraction of unharvested biomass per recruit, accounting for the age/size selectivity of the fishing gear. In the foodweb model, a balanced fishery could simply harvest a constant proportion of the production of each trophic compartment.

Consider a simple three species food chain such that $P_2 = te_2P_1$ and $P_3 = te_3P_2$. Harvesting 20% of the production of each species would decrease P_1 by 20%, P_2 by 36%, and P_3 by 49%, because there is no compensation and any reduction in producers is propagated to consumers. The only way to reduce the production of all species by 20% would be to harvest only species 1. In real communities, density-dependence can compensate, to a finite extent, for the harvest of each species but there is still a reduction in the production passed on to consumer groups. The same effect is seen in balanced harvesting of size-structured models, whereby the largest species decline more than smaller species.

The case of the Barents Sea illustrates that modelling to achieve balanced fishing should not be overly simplified and should include some species biology, not just size considerations. The wide variations in F among species harvested make fixed F not representative of actual species-related F. Preliminary analyses by the Institute of Marine Research in Norway have examined the short-term effects on the current fishery, where the TAC on cod and capelin are weighted against the strength of the cohorts and the demand for prey by the NEA cod stocks and the sea mammals. Capelin spawn and then die; reducing F does not make the fish live longer. Capelin also have extremely variable recruitment. An average F set only based on balanced fishing considerations would not allow to take account of the large fluctuations in capelin biomass and may miss high catches in good years, but harm recruitment in poor years.

5.3.2 How could any determined fishing regime be implemented?

Since the definition of "balanced harvesting" relies upon a theoretical view of the ecosystem, its implementation may be assumed to require an "ecosystem engineering" approach. The first step would consist in estimating the productivity of each ecosystem component. Then some ecosystem model would be used to derive the corresponding "balanced" mortality rates, and prescribe the authorized catches per species and size class to achieve this mortality rates. The appropriate gear and fishing effort combination to extract these catches would then be determined. This is a selective and prescriptive approach to implementing "balanced harvesting" which would

rely on the availability of appropriate ecosystem models, and gear selection curves for all harvested species.

However, there may be alternative approaches to achieve "balanced harvesting" or any agreed distribution of fishing mortality across ecosystem components that would be deemed desirable, be it "balanced" or not. Such approaches would start from the desired outcome; a "balanced" ecosystem with all components of the marine food-web in harmonious proportions. This desired state could be expressed as targets for high level indicators at the ecosystem level, such as for example the size spectrum slope or relative proportions of guilds. A diversity of gears targeting a diversity of species and size ranges would be allowed to be deployed, and the indicator(s) would be monitored. Management would then use corrections to the gear combination to get closer to the target(s), by devising *ad hoc* measures such as reduced fishing pressure by a given fleet and/or on given species or size classes.

5.4 Ecosystem consequences of fishing regimes

Consequences of balanced harvesting have been contrasted to various patterns of what has been coined "selective" fishing and several studies rather contrasted various "selectivity" regimes without including an actually balanced pattern. This section reviews and examines the consequences of various exploitation patterns, or fishing regimes, defined at the community level, beyond just balanced harvesting.

5.4.1 A review of model predictions on the ecosystem consequences of contrasted fishing regimes

A number of models have now been used to predict the ecosystem consequences, and yield benefits, of fishing regimes with different "selectivities". Model conclusions differ, probably owing to differences in model assumptions about key processes in the foodwebs, and in the way selective and/or balanced fishing were modelled. Model details are summarized in Table 5.3.

5.4.1.1 Size spectrum models

Size spectrum models have been used to investigate properties of fish communities related to the concept of balanced harvesting. The size spectrum models provide a convenient approach to balanced harvesting, as they include indirect effects of fishing and the possibility to include immature individuals into the fishery. The size spectrum modelling approach to fisheries impact is generic. Size spectrum approaches to balanced harvesting include simulations of fish communities with identical life histories (Law *et al.*, 2012; Rochet and Benoît, 2012), differentiated size dependent life histories (Jacobsen *et al.*, 2014) or actual descriptions of fish species (Hintzen *et al.*, in prep).

Studies investigating the impact of balanced harvesting on a single population, yield, community structure and resilience concluded that balanced harvesting performed better for all metrics (Law *et al.*, 2012; 2013). The balanced fishery, where fishing mortality is scaled with the productivity, is predicted to give a large total yield increase (≈50%) (Law *et al.*, 2013). The fishing patterns that scale F with size dependent productivity are contrasted to size-at-entry fisheries, where only a range of large individuals are targeted. Rochet and Benoît (2012) did not explicitly investigate balanced harvesting, but found that fishing-generated oscillations in the biomass flow appear at lower fishing intensity and have wider amplitude when fishing is selective

(narrow variants of dome shaped selection curves) and/or when large fish are targeted, than when the fishing mortality is more broadly distributed.

At the WGECO meeting a study using a size-based model of fish communities was presented (Jacobsen *et al.*, 2014). In this study the term "balanced" harvesting was used for a fishing mortality scaled with pseudo-species and size-specific productivity, defined by scaling relationships nested in metabolic theory. Furthermore the study divided fishing mortality into "selective" or "unselective" fisheries, denoting whether or not juvenile fishes were included in the fishery. The conclusion is that unselective balanced harvesting provides a slightly larger total yield (≈10% more than selective and unbalanced fishing) in contrast to single species studies, which predicted a larger increase (Figure 5.1). Balanced harvesting also showed less impact on the size structure of the fish community, and a significant reduction in the size of fish in the catch.
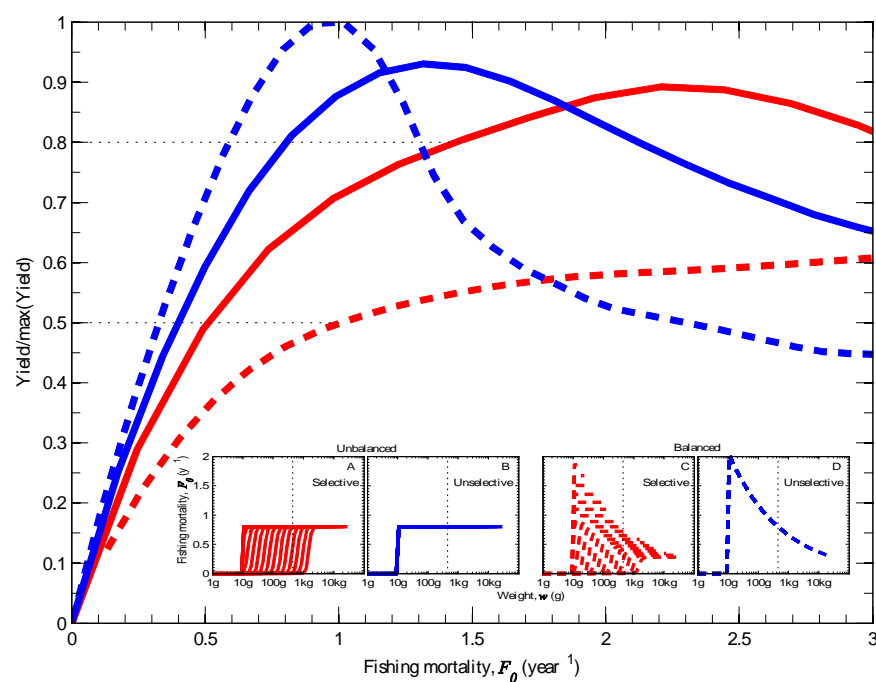
Figure 5.2. Total yield from four exploitation patterns, combining selective or balanced exploitation. The unselective balanced harvesting gives the highest total yield. Modified from Jacobsen *et al*. (2014).

In the Hintzen *et al*. (in prep.) study, the underlying dynamics of twelve fish species are calibrated to survey data and single species stock assessment results available over the period 1985–1995. The foodweb model describes the number of individuals per species available over the total size spectrum. Growth, reproduction and mortality for each species reflect available knowledge of their life-history characteristics (Blanchard *et al*., 2014). Fishing mortality can be generated by assuming an effort and a selection of a fishery. The sum of these, multiplied by a species-specific catchability, results in a fishing mortality which directly affects the survival of the fish per weight bin. The balanced fishing regime is compared to a 'Business As Usual' (BAU) scenario based on the management objectives currently in place for these species (ICES Advice, 2012).

Results suggest that balanced fishing would result in markedly higher SSB for several species (Table 5.1), slightly more relatively large fish in the community (higher LFI) but less long-lived species (lower mean maximum length) than under BAU (Table 5.2). Despite these higher SSBs, balanced fishing estimated catches were lower, possibly because single-species F_{MSY} values were used in this study. Simulations suggest that it might be necessary to introduce an intermediate state to move from one type of selection to another type of selection to prevent possible depletion of any of the stocks as the ecosystem is adjusting to the new exploitation regime.

Table 5.1. Change in SSB (%) per species after 50 years of balanced fishing compared to BAU scenario. A scenario without fishing is provided for reference.

SPECIES	NO FISHING	BALANCED FISHING
Sprat	159	85
Sandeel	301	262
N.pout	186	143
Herring	-3	-1
Dab	142	136
Whiting	1	-12
Sole	28	32
Plaice	18	1
Haddock	-36	-24
Cod	480	456
Saithe	60	-91

Table 5.2. Change in indicator value (%) after 50 years of balanced fishing compared to BAU scenario. A scenario without fishing is provided for reference.

INDICATOR	NO FISHING	BALANCED FISHING
LFI	19	5
Mean maximum weight	2	-8

5.4.1.2 Other models

Using the length-based multispecies model LeMans, Rochet *et al*. (2011) found that neither selective nor balanced fishing with respect to size can be said to be generally preferable for conserving biodiversity. The outcome depended on both the particular species composition and size structure of the community, and the shape of the selectivity function (both sigmoid and dome-shaped selection curves were used). Also, catching a narrow range of species almost always reduced evenness and species richness more than taking the same catch from a broader range of species.

Based on the simulation of end-to-end models (EcoPath or Atlantis) parameterized for 36 ecosystems, Garcia *et al*. (2012) found that unselective (including "balanced") fishing patterns allowed higher total system yields than more selective ones (partly because more components were targeted), with lower predicted impacts on the ecosystems, such as population extirpations (local extinctions) and biomass depletion (see Table 5.3 for details).

5.4.1.3 Summary

In summary model results about the consequences of various fishing regimes are nuanced. A very broad summary would be that less selective (including balanced) fishing regimes would produce higher yields with lower ecosystem impacts, but with a high variability of effect sizes, and even sometimes contradictory effects, both within and among studies. Ultimately, the differences in yield and/or impact cannot be said to be generally of significant magnitude. The consequences may depend on the combination of the settings (structure and functioning) of a given community, and the details of the fishing regime.

Moreover, studies looking at balanced fishing have often focused on yield as aspects of MSY, particularly at the fisheries objective to obtain the maximum total biomass across species and size groups (Jacobsen *et al.*, 2014). It should be noted, however, that the maximum total biomass yield does not necessarily lead to the greatest benefit to fisheries (Gislason, 1999; Rindorf *et al.*, 2012), and also may have serious predicted effects on the persistence of sensitive species in the ecosystem when using alternative models (Gislason, 1999; Smith *et al.*, 2011). Hence, any further testing of balanced fishing scenarios should relate to other versions of MSY than the aggregate biomass yield of the system; issues such as precautionarity in the exploitation of individual species must be considered whenever possible.

Table 5.3. Community models used for strategic assessment of fishing patterns, and their properties relevant to balanced harvesting.

MODEL/ATTRIBUTE	SIZE STRUCTRE	TROPHIC INTERACTIONS	ALTERNATIVE FISHING PATTERNS INVESTIGATED	RECRUITMENT	PRODUCTIVITY	BIODIVERSITY
Jacobsen *et al.*, 2014	Continuous size classes	Size based foraging	Balanced scaled with production, selective/unselective in terms of juveniles/adults	Beverton–Holt like/energy intake dependent	Emergent property of food and mortality	Size and life-history diversity dependent on maximum asymptotic size (fish community)
Benoît and Rochet	Continuous size classes	Size based foraging	Dome shaped with various targets = modal sizes, and size selectivity = inverse dome width	Linear dependence on standing stock/ size-varying fecundity	Emergent property of food and mortality	Fish, one life history with random variants
Blanchard *et al.*, SBM	Continuous size classes	Size based foraging and species- based coupling strengths	Balanced, flat unselective,	Beverton–Holt like/energy intake dependent	Emergent property of food and mortality	commercially important fish species for the North Sea
LeMans **Rochet *et al.*, 2011**	Continuous size classes	Size based foraging and spatial overlap	Dome shaped with various targets = modal sizes, and size selectivity = inverse dome width / S-shaped with varying steepness and L50.	Ricker	von Bertalanffy growth, mortality emergent	fish species for Georges bank and North Sea fish communities
Law model	Continuous size classes	Size based foraging	Balanced, flat, size selective	Linear dependent on food intake	Emergent property of food and cannibalism	One fish species
EwE **Garcia *et al.***	No, possibility of stanzas (splitting species into life cycle stages?)	Relative species composition in diet	Conventionnally selective = all groups historically fished; Unselective = all groups that have been fished somewhere in the world; Balanced No size-selection	No recruitment per se, Compartment increases with energy influx	Subject to Ecopath input (P/B) and vulnerability dynamics	End-to-end model with *n* possible species or functional groups
ATLANTIS **Garcia *et al.***	ten age/size classes for all vertebrates	Diet matrix	Conventionnally selective = all groups historically fished; Unselective = all groups that have been fished somewhere in the world; Balanced No size-selection	Beverton–Holt (food dependent)	Emergent	End-to-end model with functional groups

5.4.2 A review of the available evidence of ecosystem consequences of fishing regimes

To provide empirical evidence of ecosystem consequences of fishing regimes, first, metrics describing the selectivity or concentration of fishing across ecosystem components must be developed.

5.4.2.1 Measuring the degree to which existing fisheries are balanced

A number of metrics have been proposed for fishing pressure, describing, in addition to fishing intensity, how fishing pressure is apportioned across species and sizes (Fauconnet *et al.*, in review; Collie *et al.*, 2013; Rochet *et al.*, 2013 a,b, Table 5.4).

Table 5.4. Metrics of fishing pressure to measure the intensity and "selectivity", or degree of concentration, of fishing pressure on community components.

SOURCE OF INFORMATION / TYPE OF METRIC	STOCK ASSESSMENTS	CATCH STATISTICS
Fishing intensity	Average F* across species	- total catch weight per surface area
Fishing selectivity wrt length	SD[†](F) across length classes	- length range of catch
Fishing selectivity wrt species	SD(F) across species	- number of species that make up a given (high, e.g. 85%) proportion of total catch - percent total catch accounted for by the two most caught species - catch species richness - catch species evenness
Fishing target		- percent total catch from species groups, e.g. predators, or other functional groups - exploitation index[‡] per species group - catch average length

*F: fishing mortality rate †SD standard deviation ‡exploitation index: ratio of landings summed across species within groups to a group biomass index from e.g. a survey.

5.4.2.2 Contrasting the ecosystem effects of fishing regimes

Temperate shelf fish communities have been heavily exploited, but many experienced decreasing fishing pressure and changes in selectivity in the most recent decade. Several studies have examined the consequences of these changes.

Collie *et al.* (2013) compared community rebuilding on Georges Bank and in the North Sea. Metrics of overall fishing mortality and the degree of species and size selection at the community level were developed from stock assessment results. The faster rebuilding rate on Georges Bank compared to the North Sea can be ascribed to several causes which were probably combined. The communities were different to start with, and underwent different environmental changes. The changes in fishing pressure were more gradual in the North Sea; on Georges Bank, the decrease in fishing intensity was accompanied by a decrease in fishing selectivity, which may have accelerated the rebuilding of the target species, and thus of the community itself.

The consequences of fishing selectivity in communities has also been investigated by taking a functional group approach applied to three North Atlantic fish communities, Georges Bank, the Bay of Biscay, and the North Sea (Rochet *et al.*, 2013 b). The three communities underwent fisheries with contrasted selectivities both across and within functional groups; fishing pressure changed substantially in all three communities over the time periods examined. Community responses to perturbation were mostly determined by (i) community structure and (ii) whether pressures were synergistic or antagonistic. The propagation of antagonistic pressures potentially generated by less concentrated fishing regimes is less predictable than simpler perturbations. For example, harvesting both prey and predator groups, releases natural mortality on the prey while exerting fishing mortality, creating antagonistic pressures on this group. Hence, since more balanced fisheries tend to create multiple pressures on exploited communities, they may be more difficult to manage.

Another study investigated the link between fisheries selectivity and biodiversity in exploited communities by a comparative analysis of fishing pressure and impact across a range of temperate exploited shelf communities (Rochet *et al.*, 2013 a). In this analysis individuals were "ecological units;" distinct communities with defined fishing patterns. The study sought evidence of a link between metrics of fishing pressure, including selectivity metrics, and metrics of fishing impacts. Time-series of fishing pressure metrics (FPMs, based on catch statistics) and fishing impact metrics (FIMs, based on bottom-trawl survey data) were calculated from 13 temperate shelf sea communities from the western and eastern North Atlantic and the Mediterranean. FPMs were averaged over ~10 year time periods with consistent levels of fishing pressure; FIMs were averaged across the subsequent ten year period, allowing for a ten year lag between pressure and impact. The relationship between pressure and impact metrics was examined by a canonical correlation analysis. Although stark contrast was found between FPMs across places and/or time periods, only a weak link (22% of total variance) was found between fishing selectivity and the community biodiversity ten years later.

Overall these studies provide scarce evidence that fishing patterns would directly determine community structure and biodiversity. Exploitation patterns may affect the community responses to changes in the environment or fishing intensity; but the size of these effects is not such that a strong signal can be detected among the noise of the many other factors.

5.5 Conclusions

5.5.1 Consequences for management advice

Model predictions of the consequences of balanced fishing or other fishing regimes are nuanced; evidence of these consequences from real systems is scarce and weak. The intuition that more balanced fishing patterns might alleviate ecosystem impacts of fisheries cannot be said to be substantiated so far. Also, balanced fishing patterns may be difficult to implement, both because they may result in less predictable ecosystem dynamics, and owing to the complexity of translating the concept into practical management measures. The message to management bodies aiming at implementing an ecosystem approach to fisheries can only be mild at this point; it may be precautionary to avoid too selective fisheries, but whether a balanced exploitation should be aimed at remains an open question. Whereas size-selective fishing seems to have a demonstrated impact on populations, fishing regimes with respect to size at the community level might not be as important an issue.

Alternatively, balanced fishing may be viewed as a thought experiment rather than an operational objective. It can be seen as the extreme end of a continuum of exploitation patterns, the other end of which would be a perfectly selective fishing which would remove only the most valuable species. The concept is primarily an incentive to think of exploitation patterns at the community or ecosystem level and their likely consequences; a consideration that should be a key component in developing an ecosystem approach to fisheries.

5.5.2 Balanced fishing and the landing obligation in the new European Common Fisheries Policy

"Balanced fishing" may be at odds with the landing obligation, although the consequences of the landing obligation are largely unpredictable. In general, the landing obligation (LO) can be seen as leading to more rather than less selective fishing. There are two approaches that might be taken to the LO. The first would be to minimize the unwanted catches, the second would be to make use of the unwanted and unavoidable bycatch. Minimizing the unwanted catch, will almost certainly lead to more selective fishing, most obviously in terms of avoiding catches of small fish. This is directly in opposition to the "balanced fishing" concept. It will also likely include avoidance of "choke species" which are one of the main causes of discarding of commercial species. Again, this would be expected to lead to more selective fishing, in this case, species-selective fishing. The second approach, to land unwanted and unavoidable catch and make use of these, is quite different. This approach can probably be seen as in line with balanced fishing. Indeed, Garcia *et al.* (2012) suggested that there will be a need to develop markets and handling systems for previously unwanted catches, e.g. small fish. So the key issue with the LO is how the balance between these two approaches evolves. In some fisheries and regions the balance may go towards avoidance, while in others towards making use of these catches. It is likely that in most cases the response will be a combination of the two approaches.

5.5.3 Research suggestions

So far modelling studies of balanced fishing have relied mostly on size-based approaches. Other complementary approaches might be useful to investigate the concept further. Important management aspects like conservation of vulnerable species as well as consideration of fishing impacts on benthic habitats should be further investigated in more detail within the concept of balanced harvesting.

Along the lines suggested in Section 5.3.1 above, simple theoretical (as opposed to numerical) models could be used to examine the very broad conditions required for balanced harvesting to be possible. The balanced harvesting concept may not be generally practicable for simple production and energetic reasons; this question requires further investigation.

The balanced *vs.* concentrated fishing regimes could also be investigated more thoroughly with respect to dimensions other than size. For example, the consequences of extirpating a species from a trophic guild, or an entire trophic guild (if it is possible), are of interest as selective fishery policies develop.

Most modelling work on "balanced fishing" has concentrated on analysing "perfect" balanced fishing. However, it is generally agreed that achieving "perfect" balanced fishing in reality will be quite difficult. So an appropriate research question might be: "Does partial progress towards balanced fishing yield at least some of the benefits expected of full balanced fishing?" This should be fairly easily realized by the same

modelling approaches already deployed to examine the consequence of balanced fishing.

On the empirical side, broader scale analyses of the actual fishing regimes would be useful. For example, establishing how balanced is fishing e.g. in the North Sea is an interesting question.

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6 ToR d) Work towards including new research on reducing effects on the seabed and associated communities of fishing operations and gears, including ghost fishing in ecosystem advice

6.1 Introduction

Bottom trawling has occurred in many decades and even centuries (Smith, 1994; Engelhard, 2008) and has affected large areas of the continental shelf seabed in Europe and elsewhere around the world (Rijnsdorp et al., 1998; Pitcher et al., 2000; Roberts, 2007). Fishing gear affects seabed habitats and kills or injures benthos. The spatial extent of the impact of fisheries has increased over time due to technological innovations (such as rock-hopper gear and chain mat beam trawls) and the increase in size and power of fishing vessels and their gear. This combined with developments in GPS plotters and echosounders has allowed bottom-trawl fisheries to extend their activities into previously untrawlable grounds (Morato et al., 2006). The many years of bottom trawling is likely to have caused structural changes in benthic habitats by altering sediment structure or removing biogenic structures such as corals or biogenic reefs (Roberts, 2007). Experimental trawling was conducted in the fisheries protected zone near Bear Island, in an area that has been closed for trawling since 1978, in the Barents Sea, to study immediate effects of otter trawling on the substrata and the benthic assemblage (Kutti et al., 2005). The substrata in the trawled sections became softer while the biodiversity increased. Several long-term studies have shown changes in the benthos, in particular the decrease of long-lived slow-growing species and the increase in short-lived fast growing species (Pitcher et al., 2000; Tillin et al., 2006). The interpretation however is not unequivocal since some of the observed changes could also be caused by pollution or climate change (Borja et al., 2000; Kroncke et al., 2011). Norwegian and Mediterranean studies on otter trawling impact (Smith et al., 2000; Lucchetti et al., 2011; Buhl-Mortensen et al., 2013) showed that trawling imposed a functional change on the megafaunal community structure, where sessile or discretely mobile filter-feeding organisms are replaced by mobile scavengers and opportunists. The ecosystem effects related to the use of bottom gear may extend far beyond the direct impacts discussed above. For example, eutrophic processes in closed basins and low depth (as in the northern Adriatic) may be enhanced by trawling, leading to hypoxia in sensitive soft bottom areas and an increase in the quantity of hydrogen sulphide released from sediments (Caddy, 2000; Lucchetti et al., 2011). A further complication for the appropriate assessment of the impact of bottom trawling is the lack of suitable untrawled reference areas (Løkkeborg, 2005). Few studies have been able to compare the benthos between untrawled reference areas and trawled areas (Blyth-Skyrme et al., 2004). For instance, Duineveld et al. (2007) showed the higher abundance of habitat engineering species in the safety zone around oil platforms in the intensively trawled southern North Sea. In addition reference areas are often not representative because they are not selected at random. Recent comparative field studies, utilizing fisheries data collected at the appropriate resolution, suggested that benthic biomass decreased with increasing trawling frequency (Hinz et al., 2009; Jennings et al., 2001). These studies, however, do not provide insight into the underlying mechanisms. In general, our poor level of mechanistic understanding of benthic ecosystem state and functioning has hampered the integration of bottom fauna into ecosystem based fisheries management. For instance, there is still debate about the effectiveness of the Plaice Box, an area in the coastal waters of the southeastern North Sea that was closed to large beam trawlers to reduce the excessive discarding of un-

dersized plaice. After the establishment of the Plaice Box, discarding has not been reduced because the undersized plaice have moved to deeper waters outside the box. It is unresolved whether this is due to the lack of bottom trawling in the Plaice Box which has reduced the food availability for plaice, as fishers claim, or due to the increase in temperature (van Keeken *et al.*, 2007; Verweij *et al.*, 2010). Another problem in quantifying the impact of trawling on the benthos is the lack of data on the frequency of fishing at appropriate spatial and temporal scales. Although data on the distribution of fishing effort is available for historic periods (Jennings *et al.*, 1999; Engelhard *et al.*, 2011), the spatial resolution of the data (ICES rectangles of ~50x50 km) is too crude because fishing effort has been shown to be highly patchy at a scale of ~2x2 km (Rijnsdorp *et al.*, 1998). It is only since the introduction of the Vessel Monitoring System that fishing effort is recorded at the appropriate spatial resolution (Deng *et al.*, 2005; Murawski *et al.*, 2005; Mills *et al.*, 2007; Mullowney and Dawe, 2009; Lee *et al.*, 2010; Hintzen *et al.*, 2010; Gerritsen and Lordan, 2011). With the high-resolution VMS data of the relevant fisheries, trawling frequencies can be estimated at appropriate spatio-temporal scales for different benthic communities to assess the impact on communities of different sensitivities. In order to understand how fishing may impact benthic ecosystems, there is a need to develop a mechanistic understanding on the 'key' processes that determine the structure and functioning of the benthic ecosystem as well as having the knowledge of how fishing may impact these 'key' processes.

6.1.1 Benthic community

Benthic organisms perform a number of ecosystem level processes, often described as 'ecosystem functions'. These functions encompass the process of transformation, whether measurable or not, that occurs in an ecosystem. They include all metabolism, catabolism and dynamic processes such as sediment bioturbation or active resuspension, as well as the production and transfer of food, oxygen, and nutrients, the recycling of waste material and the sequestration of harmful substances. While some ecosystem functions can be undertaken by a variety of different organisms, it is generally considered that a greater diversity of species increases the stability and resilience of an ecosystem's capacity to perform its various functions (Cardinale *et al.*, 2000; 2002).

Linked to this hypothesis is the notion of functional redundancy in ecosystems, where the loss of a species belonging to one functional group may not affect the basic functioning of the ecosystem, when the function performed by that species is taken up by another member in the same functional group (Fonseca and Ganade, 2001). The extent to which species can be lost before basic ecosystem processes are compromised depends on the functional richness (i.e. the number of functional groups), the number of species in each functional groups and evenness (i.e. the distribution of species across functional groups) in an ecosystem (Mouillot *et al.*, 2005). In order to understand the functioning of benthic ecosystems, studies have focused on the functional role of benthic organisms, such as their trophic role (filter-feeder, deposit-feeder, predator, omnivore) or the position in the sediment (epifauna or infauna). Biological Traits Analysis or BTA (Bremner *et al.*, 2006) which uses a series of life-history, morphological and behavioural characteristics of species present in assemblages, is a powerful method to classify the ecological functioning of benthic organisms. This method also provides an opportunity to classify the sensitivity of populations of benthic organisms to the additional mortality imposed by fishing. For example, communities made up mainly of short lived fast growing species will likely be less sensitive

to trawling induced mortality as compared to populations characterized by long-lived and slower growing megafauna species. In contrast, habitat forming benthic organisms or bioengineers such as corals, sponges, oyster beds, polychaete worm reefs or crustaceans that create networks of burrows in soft sediments, need particular attention because the habitat created by them has a much longer lifespan than that of the individual organisms. As a consequence these habitats develop slowly and the recovery rate will be slow as compared to the life time of the individual. In addition the resulting biogenic habitat provides a place to live for many other species which are dependent on this habitat. The processes that determine the population dynamics of benthic organisms are driven by a balance of abiotic (e.g. physical environmental forcing) and biotic processes (e.g. competition for food and space) acting in combination through a complex network of feedback mechanisms. At one extreme, sedimentary habitats exposed to high natural disturbance caused by sediment transport via tide and wave action, results in benthic communities that are highly resilient to physical disturbance. These communities are often described as being r-selected (selected on reproduction) (Pianka, 1970). By contrast, seabed environments that are physically stable tend to favour the presence of species with large individual body size, which results in populations that tend to be resistant to initial physical perturbations, but once impacted take much longer to recover than those which are r-selected. These communities are often described as K-selected (Pianka, 1970) (competition selected). K-selected populations, composed of relatively large individuals, often provide habitat structural heterogeneity which is important for other species (Buhl-Mortensen *et al.*, 2010). They also maintain a potentially important buffer for the storage and transfer of carbon and nutrients. Benthic macrofauna play a significant role in many marine ecosystems by regulating biogeochemical processes through burrowing in the sediment and by regulating the benthic-pelagic coupling (Lohrer *et al.*, 2004). Furthermore, recent research conducted into marine foodweb dynamics has revealed the relative importance of bottom–up (resource-driven) and top–down (consumer driven) forcing in regulating the status and function of marine ecosystems, particularly in relation to the sustainability of commercial fish stocks (Frank *et al.*, 2007; Kenny *et al.*, 2009, Section 8 in this report).

6.1.2 Fishing impact

Fishing can affect benthic ecosystems in many ways, by modifying the sedimentary habitats, increasing or decreasing nutrient fluxes, killing benthic invertebrates and through the redirection of energy via discards to the seabed. These changes in turn lead to changes in the functioning of the benthic ecosystem and the availability of food for commercial fish species. The different pathways by which fishing may impact the benthic ecosystem are summarized in Table 6.1 distinguishing between the mechanism and the ecological effect, and are discussed below. Commercial fisheries utilize a wide variety of fishing gears ranging from passive gears such as pots and trammelnets, to bottom trawls that are towed over the seabed. Passive gears may damage benthos, for instance when a longline deployed on a reef may tear off branches of the reef, but it is generally assumed that bottom trawls will have a much larger impact on benthic ecosystems than passive gear because the footprint of towed gears is many orders of magnitudes larger than those of passive gears thereby causing greater mortality rates of benthos and higher habitat modification rates (Jennings and Kaiser, 1998). The impact of a bottom trawl will depend on the size, weight and design of the gear components, their penetration depth as well as the speed and distance over which the gear is towed. For example, in an otter trawl, the sweeps only touch the surface of the seabed, whereas the otter boards dig a furrow into the sedi-

ment. Many trawl doors are the result of initial designs, improved through practical trials until they work well enough to be used commercially. Modern door designs are more advanced and sophisticated as a result of increasing fuel costs and the necessity to minimize impact on the environment. Meeting these challenges has led to significant improvements in the way new otter boards are designed and tested (Sala *et al.*, 2009). In a beam trawl, the tickler chains mounted between the shoes penetrate into the sediment and disturb the upper layer as well the benthic organisms that live in the sediment. The penetration depth depends on the number of tickler chains and on the sediment type (Ivanovic *et al.*, 2011). A promising development is the numerical modelling of the physical impact of a fishing gear on the seabed based on the characteristics of the gear. Ivanovic *et al.* (2010) have developed such a model for an otter trawl distinguishing between different components such as the otter door and the roller clump and validated the model in sea trials on two sediment types where the physical alteration to the seabed following the passage of a roller clump and a trawl door was measured and profiled. If extended to other gear components, and thoroughly validated, this approach offers great potential to predict the physical impact on the seabed of a variety of gears in different benthic habitats without carrying out physical experiments.

The disturbance of the sediment may cause changes in the geo-chemical processes in the seafloor (Duplisea *et al.*, 2001). Resuspension of organic material (Durrieu De Madron *et al.*, 2005; Pilskaln *et al.*, 1998) may affect the nutrient and carbon fluxes from the sediment, and consequently affect primary production and eutrophication. O'Neill and Summerbell (2011) have demonstrated that, for a given sediment type, there is a relationship between the hydrodynamic drag of the gear element and the mass of sediment entrained behind it.

The direct mortality imposed on organisms that are hit by a fishing gear has been estimated in field experiments. Mortalities vary between species, fishing gears and sediment type but are generally around 50% for a single passage of a trawl (Kaiser *et al.*, 2006). Beam trawls and scallop dredges on average cause greater mortality rates than an otter trawl. Biogenic habitat building species are more vulnerable than infaunal invertebrates. There is a large body of evidence on which predictions based on sediment type, gear, and taxononmy can be made for fishing mortality on benthic species. However, many benthic organisms and bottom-trawl gears remain, for which no direct mortality estimates are available (Kaiser *et al.*, 2006).

Table 6.1. The different mechanisms through which fishing may impact the seafloor. The pressures are according to the MSFD.

MECHANISM	MSFD PRESSURE	STATE: ECOSYSTEM COMPONENT	STATE ASPECTS IMPACTED
Direct effects through extraction of shellfish	Biological extraction	Benthos	Abundance/Biomass Productivity Structure: Size
Indirect effects of removal of fish through predator–prey relationships			Taxa Functional groups
Destruction of structural elements	Abrasion Habitat loss	Habitat: Hard structures	Various habitat functions
Disturbance of sediment (e.g. Homogenization, Compression, Resuspension)	Habitat loss Smothering Siltation Nitrogen & Phosphorus enrichment	Habitat: Soft substrata	Various habitat functions
Trawl path mortality	Abrasion Smothering	Benthos	Abundance/Biomass Productivity Structure: Size Taxa Functional groups
Food subsidies through discarding	Input of organic matter	Benthos	

The modification of the seabed habitat, mortality of invertebrates and flow of discards to the seabed has resulted in long-term changes to the functioning of benthic ecosystems. Fishing results in changes in the species and size composition of the benthic community due to differential mortality across species and size classes, and due to the food subsidies provided by the trawl track mortality and the discards and offal that sink to the seabed. Community changes will influence the ecosystem functioning affecting geo-chemical fluxes as well as trophic interactions (Dayton *et al.*, 1995; Kaiser *et al.*, 2000; Tillin *et al.*, 2006). Few studies have been carried out to evaluate how the effects of large-scale commercial fisheries results in geo-chemical and community changes and how these translate into effects on measures of ecosystem functioning such as bioturbation, nutrient fluxes and bentho-pelagic coupling. Furthermore, we currently lack the ability to evaluate the effect of large-scale chronic trawling on the food availability for benthivorous fish such as plaice, cod and haddock. Such changes in food availability may affect secondary production in fisheries (Hiddink *et al.*, 2011). Little is known on how fishing indirectly affects bioturbation, nutrients fluxes and bentho-pelagic coupling through changes in benthic community composition. Trawling has been shown to reduce the abundance of bioturbating species and this is likely to affect nutrient fluxes (Widdicombe *et al.*, 2004). Trimmer *et al.* (2005) found that biogeochemical processes in the upper layers of sediment, both oxic and suboxic,

seemed unaffected by trawling in the long term. In deeper anoxic sediment however, mineralization via sulphate reduction may be stimulated by the extra disturbance, at least in areas where tidal energy is slight. Studies on the effect of trawling on the bentho-pelagic coupling have so far not been conducted. A serious deficiency of our understanding and predictive ability of the effect of trawls on ecosystem functioning therefore remains. Many important commercial fishes, such as flatfish and gadoids, feed on benthic invertebrates for part of or all their life history. Bottom trawling thus not only reduces the population size of fish through direct removal, but also reduces the abundance of their prey (Auster and Langton, 1999). Recent studies have shown that this may results in reduction of the growth of flatfish species (Hiddink *et al.*, in press; Shephard *et al.*, 2010), and could therefore reduce the sustainability of fisheries. These results contrast to the hypothesis that bottom trawling may promote the typical small benthic organisms on which small-mouthed flatfish species like sole and plaice feed (Hiddink *et al.*, 2008; Rijnsdorp and Van Beek, 1991; Rijnsdorp and Van Leeuwen, 1996). We are currently lacking the ability to assess to what extent such trawling induced changes in food availability are affecting fisheries over large scales and for most important fished species. Only a few studies have attempted to model the large-scale effects of chronic trawling impacts on the benthic ecosystem structure and functioning. Duplisea *et al.* (2002) and Hiddink *et al.* (2006) used a size-based model to show that current bottom-trawl activities in the North Sea resulted in a 56% reduction in biomass and 21% reduction in production of benthic invertebrates in the southern North Sea. This model was applied to address the question how the reduction in beam trawling in the Plaice Box, an MPA established to reduce the bycatch of undersized plaice, could have affected the food for plaice that feed on small benthic invertebrates (Hiddink *et al.*, 2008). It was shown that the overall biomass and production of the benthic ecosystem decreased with increasing trawling intensity, but that the production of suitable prey, small worms, was low without trawling and maximal in areas that are trawled once to twice a year, suggesting that the food for plaice may have been reduced within the Plaice Box following the reduction in beam trawling in the box. Allen and Clarke (2007) used a coupled physical-ecological model (the European Regional Seas Ecosystem Model (ERSEM) with the General Ocean Turbulence Model (GOTM)) to investigate the impact of demersal trawling on the benthic and pelagic ecosystems of generic stratified and unstratified water columns in the central North Sea. The modelling suggests that the biogeochemical impact of demersal trawling is most significant in regions where the gear type, trawl frequency and bed type cause high levels of filter-feeder mortality. This results in significant changes in its biogeochemistry (increased phosphorus absorption, increased nitrification of ammonia, reduced silicate cycling). Our ability to predict the ecosystem effects of fishing at a regional scale requires sophisticated models, and is therefore currently hampered by insufficient knowledge of how fishing affects different ecosystem functions in different habitats. Furthermore, to allow meaningful management of these fisheries, the effect of fishing relative to the natural variations such as those caused by storms must be known.

6.1.3 Ecosystem approach to fisheries management

The knowledge of the impact of fishing on the structure, functioning and services of the benthic ecosystem as well as the socio-economic consequences of management measures is the scientific basis required to develop an ecosystem approach to fisheries management (EAFM) which is required according to the main marine policy directives: Common Fisheries Policy and Marine Strategy Framework Directive as the ecological part of the Integrated Maritime Policy. As there are many different mecha-

nisms through which fishing impacts seafloor integrity possibly affecting the services it provides, the first requirement to assess the impact of fishing and mitigate this through EAFM is a comprehensive suite of indicators covering the different pressures as well as all the relevant aspects of state that may be impacted by these pressures.

6.2 Proposed new research

The previous section highlights the need to improve our mechanistic understanding of the direct and indirect effects of fishing on benthic communities in order to develop predictive models on the interaction between fisheries and benthic ecosystems. The FP7 project BENTHIS (http://www.benthis.eu/en/benthis.htm) has taken up this challenge to strengthen the scientific basis for integrating benthic ecosystems into fisheries management. Below we highlight some specific topics of this project to direct initiatives for new research.

6.2.1 Biological traits analyses

Traditional studies of the status of marine benthic habitats have relied on the diversity, abundance and biomass of taxa, typically at the species level. However, changes in taxonomic or systematic diversity do not necessarily reflect changes in habitat functions or the provision of ecosystem goods and services associated with these functions, such as those related to bioturbation or the provision of structural habitats for protection or spawning. Biological traits analyses (BTA) (e.g. Bremner *et al.*, 2006; Bolam *et al.*, 2013) use a range of life-history, morphological and behavioural characteristics of the organisms as indicators of their ecological functioning. It can thus be viewed as a measure of functional diversity. One task taken up by BENTHIS is to identify and quantify the most relevant and significant relationships between biological traits of infauna and epifauna genera and their functions (e.g. ecosystem services they provide) for a range of habitat types (subjected to varying amounts of natural disturbance) representative of European Regional Seas (Table 6.2). In addition these genus traits will be related to important ecological functions of different habitat types and to anthropogenic pressures, including those caused by fishing.

Table 6.2. Biological trait categories for infauna genera used in analyses of changes in functional composition between different habitats types under varying level of fishing pressure in the EU project BENTHIS.

TRAIT	CATEGORIES
Morphology	Soft, Tunic, Exoskeleton, Crustose, Cushion, Stalked
Maximum body size (length) in mm	<10, 11–20, 21–100, 101–200, 201–500, >500
Longevity (maximum) year	<1, 1-<3, >3–10, >10
Larval development	Pelagic – Planktotrophic, Pelagic – Lecithotrophic, Benthic – Direct.
Egg development location	Asexuel/budding, Sexual-shed eggs-pelagic, Sexual-shed eggs-benthic, Sexual-brood eggs
Living habitat	Tube-dwelling, Burrow-dwelling, Free-living, Crevices/holes/under stones, Epi/endo-zoic/phytic, Attached
Sediment position	Surface, Infauna: 0–5 cm, Infauna: 6–10 cm, Infauna: >10 cm
Feeding mode	Suspension/filter-feeder, Surface deposit, Subsurface deposit, Scavenger/opportunist, Predator, Parasite
Mobility	Sessile, Swim, Crawl/creep/climb, Burrower
Bioturbators	Diffusive mixing, Surface deposition, Upward conveyor, Downward conveyor, None

6.2.2   Generic fishing/seabed habitat impact assessment model

In order to assess the vulnerability of benthic communities and ecosystems to the adverse impact of fisheries, a generic fishing/seabed habitat risk assessment method will be developed. Within BENTHIS, a generic fishing/seabed habitat impact assessment model was developed that integrates a number of specific tools dealing with the structure and functioning of the benthic ecosystem and the physical impact of fishing gear.

Physical Trawling impact model

Building on the work of Ivanovic *et al.* (2010), a predictive tool will be developed on the physical impact of fishing gear. The project will distinguish between the different gear components and validate the model against data collected in the BENTHIS Case Studies and will integrate this information with existing published studies where appropriate. Combining this with information on the spatial distribution of different fishing fleets will allow us to assess the extent of sediment modification by trawls over large scales. This analytical approach, in combination with the *Mortality model* below, will allow the establishment of a relationship between the physical characteristics of a gear and its direct impact on benthic organisms and habitats, by fishing gears for which no empirical measurements are available.

Resuspension model

This model will also provide data on the hydrodynamic drag of a gear element which can be used to estimate the effect on the resuspension of sediments. O'Neill and

Summerbell (2011) have demonstrated that, for a given sediment type, there is a relationship between the hydrodynamic drag of the gear element and the mass of sediment entrained behind it. Hence, a predictive model of the physical impact of fishing gear on the seabed will also provide the insight needed to quantify how fishing can affect the nutrient and carbon fluxes from the sediment, and consequently affect primary production and eutrophication, as well as destruction of benthic habitats (Riesen and Reise, 1982; Jennings *et al.*, 1998; Watling and Norse, 1998). Combining this with information on the spatial distribution of different fishing fleets will allow us to assess the extent of sediment modification by trawls over large scales.

Mortality model

Trawling impact models rely heavily on empirical estimates of trawling mortality, which are only available for a small number of fishing gears, certain habitat types and a limited range of benthic organisms. A generic tool will therefore be developed to predict the vulnerability of benthic organisms for trawling mortality by quantifying the relationship between observed mortality rates imposed by different fishing gears and biological characteristics (Kaiser *et al.*, 2006).

Ecosystem function models

Given the wide diversity of benthic organisms and their ecosystem function, the most efficient approach is to develop generic tools based on biological traits to predict the ecosystem function of benthic organisms and their sensitivity to fishing impacts. BENTHIS will study the relationship between the functional roles of benthic organisms in a variety of benthic ecosystems with biological traits, such as life-history, morphological and behavioural characteristics, that will allow us to predict ecosystem functions for organisms which have not been studied in detail (Bremner *et al.*, 2006). By combining this information with the *Mortality model* and the distribution of fishing effort, it will be possible to map the distribution of ecosystems where functioning is severely affected by direct fishing impacts.

Ecosystem functioning

BENTHIS will generate regional models to combine our understanding of the effect of chronic trawling on ecosystem functioning with spatially resolved fishing effort and fishing gear data and information on seabed content to determine the effect of fishing on bioturbation, benthic-pelagic coupling, nutrient and carbon fluxes.

Food subsidies

Studying the energy flow due to discards to the seabed is required to identify where the impact of fishing on the ecosystem is likely to change as a result of the discard ban (see Section 7 of this report).

Fish production

An ecological model of the interactions between benthos, fish and trawl fisheries will be developed based on these analyses to predict the effects of bottom trawling on fish populations and fishing yields.

6.2.3 Mapping fishing impacts

European regional seas cover a diverse range of seabed habitat types, representing a wide range of environmental conditions and community types. BENTHIS will take

advantage of recent international seabed mapping initiatives such as MESH (Mapping European Seabed Habitats) together with many national seabed mapping programmes (e.g. MAREANO (www.mareano.no)) as an invaluable source of empirical data upon which the ecological analysis can be applied. For the spatial patterns of fishing activity BENTHIS will apply (and if required further enhance) the VMS Tools library created as part of EU tender No MARE/2008/10. This allows collating national VMS data into regional maps of international fishing intensity of all relevant métiers at an appropriate spatio-temporal resolution. Maps of the fishing impacts can then be created for a number of different benthic ecosystems in the regions studied in BENTHIS by combining information of the seabed habitat types (EUNIS), with high resolution trawling frequency (VMS-based) maps for a selected number of fisheries, and the application of the Generic fishing/seabed habitat risk assessment model.

6.2.4 Ecosystem-based fisheries management

BENTHIS' contribution to (further) develop ecosystem-based fisheries management follows two approaches. Firstly, BENTHIS will test the ecological and economic performance of technical innovations currently being developed to mitigate the impact of fishing in collaboration with the fishing industry. Secondly these innovations will be considered as part of management measures and evaluated against their performance to reduce the ecological impact of the fishery and the implications it has on the economy of the fishing sector. To that end an integrated ecological and economic evaluation framework is required. This framework will then be applied to evaluate the different management measures that can mitigate fishing impact. Each measure will be evaluated against a suite of criteria that determine its performance in terms of achieving specific policy objectives on the state of the ecosystem and sustainable fisheries. In order to take all three pillars of sustainability into consideration when selecting the most appropriate management measures, BENTHIS will develop and apply an evaluation framework that applies methods of preference modelling (i.e. Multi-Criteria Decision Analysis) that combine decision-makers' preferences and value judgments on the suite of criteria with outputs from the BENTHIS impact assessment models that quantify these criteria. For the evaluation of these management measures BENTHIS will apply Management Strategy Evaluation (MSE). However, where in the European context this has mostly been applied in single-species management towards achieving Common Fisheries Policy (CFP) objectives, BENTHIS will develop the evaluation framework such that it can be applied in management involving multiple species encompassing different ecosystem components (i.e. fish, benthos, habitats) so that the performance of management measures can also be evaluated against the relevant MSFD descriptors (i.e. seafloor integrity but also foodweb functioning and biodiversity). Two focal points of research are considered to contribute to the development of an ecosystem approach to fisheries management aimed at reducing the impact on the seafloor: (1) Management measures and (2) Indicators.

6.2.4.1 Management measures

A typology was developed to identify the possible management measures and the mechanisms through which they can be implemented (Table 6.3).

Table 6.3. Typology of management measures, based on van Vliet and Dubbink (1999), Gray (2005) and de Vos *et al.* (2013), and a hierarchy of specific types of measures. Each category can be applied to create the incentives for implementation of any type of management measures.

CATEGORY	INCENTIVES	TYPE
Regulatory (Hierarchical)	Coercive	Area and/or time restrictions, Marine Protected Areas/Closed areas Zoning Real time closures
Economic (Market based)	Financial	Technical measures: modification of gear gear substitution Catch/landing restrictions
Social (Participatory)	Social/moral	TAC/Quota/ITQs Minimum landing size Discard ban Effort management Days-at-sea Public awareness Labelling schemes Environmental stewardship (MSC) Fishers behaviour Reduce footprint

6.2.4.2 Indicators

In order to assess the state of the seafloor, the pressure of the fishery and its impact on the integrity of the seafloor and evaluate the performance of management measures to reduce this impact we developed an approach that allows the selection of a comprehensive suite of operational indicators. Ultimately the selection of which of these indicators can be made operational in a particular (sub)region depends on the availability of data.

The DPSIR framework is often applied to understand the causal relationships that determine the effects of human activities on the environment and how this can be mitigated through management. The selection of indicators to assess the impact of fishing on the seafloor is therefore based on this framework and the different pathways through which fishing may impact the seafloor are given in Table 6.1. The three pressure categories distinguished in table 6.1 with its proposed indicators are given in Table 6.4.

Table 6.4. Proposed operational indicators for three fishing pressures. Pressure indicators should be reported per métier and aggregated across métiers (i.e. total).

PRESSURE CATEGORY	PROPOSED INDICATOR
Biological extraction	Catch per year
	Landings per year
Abrasion/smothering/habitat loss	Frequency of disturbance per unit area
	Frequency of disturbance at depth per unit area
Discarding	Amount of discards per year returned to the sea

Table 6.5. Proposed operational indicators for the relevant aspects of state of the seafloor. This is based on the criteria for the MSFD seafloor integrity descriptor. Where possible these indicators should be reported per predominant habitat (see Table 6.4) and aggregated across the total MSFD (sub)region.

STATE CATEGORY	MSFD CRITERIA	MSFD INDICATORS	PROPOSED OPERATIONAL INDICATOR
Physical habitat	6.1 Physical damage, having regard to substratum characteristics	Type, abundance, biomass and areal extent of relevant biogenic substrata (6.1.1)	Areal extent of biogenic reefs as described under the Habitats Directive (1170) Reefs
		Extent of the seabed significantly affected by human activities for the different substratum types (6.1.2)	DCF indicator: Areas not impacted by mobile bottom gears
Associated benthic community	6.2 Condition of benthic community	Presence of particularly sensitive and/or tolerant species (6.2.1)	Identification of sensitive/tolerant species in relation to two aspects of vulnerability based on traits (single or combination): Direct mortality and Recovery potential
		Multimetric indices assessing benthic community condition and functionality, such as species diversity and richness, proportion of opportunistic to sensitive species (6.2.2)	Taxonomic (genus level) and functional (traits) diversity and richness
		Proportion of biomass or number of individuals above some specified length/size (6.2.3)	Mean size based on biomass per Maximum bodysize trait category
		Parameters describing the characteristics (shape, slope and intercept) of the size spectrum of the benthic community (6.2.4)	Biomass per specific Maximum bodysize trait category or group of categories (e.g. largest category)

Table 6.6. Predominant seabed habitat types, including their biological communities (angiosperms, macroalgae, bottom fauna) according to the MSFD (EC, 2008).

PREDOMINANT HABITATS
Littoral rock and biogenic reef
Littoral sediment
Shallow sublittoral rock and biogenic reef
Shallow sublittoral coarse sediment
Shallow sublittoral sand
Shallow sublittoral mud
Shallow sublittoral mixed sediment
Shelf sublittoral rock and biogenic reef
Shelf sublittoral coarse sediment
Shelf sublittoral sand
Shelf sublittoral mud
Shelf sublittoral mixed sediment
Upper bathyal rock and biogenic reef
Upper bathyal sediment
Lower bathyal rock and biogenic reef
Lower bathyal sediment
Abyssal rock and biogenic reef
Abyssal sediment

6.2.4.3 Using the ecosystem services framework to guide the selection of indicators

In order to guide the selection of operational indicators covering the most relevant aspects of state that are impacted, WGECO considered the ecosystem services the benthic habitat and its associated communities is expected to provide. The most authoritative source for the classification of ecosystem services, the Common International Classification of Ecosystem Services (CICES) and its most recent version, i.e. CICES 4.3, were used. For each potential ecosystem service that may be affected by the fishing impact on the seafloor we propose one or more of what can be considered the most appropriate indicators (Table 6.7).

Table 6.7. Selection of ecosystem services based on CICES 4.3 that can be provided by the seafloor and their preferred indicators.

SECTION	DIVISION	GROUP	CLASS	CLASS TYPE	INDICATOR	EXAMPLES
This column lists the three main categories of ecosystem services	This column divides section categories into main types of output or process.	The group level splits division categories by biological, physical or cultural type or process.	The class level provides a further subdivision of group categories into biological or material outputs and biophysical and cultural processes that can be linked back to concrete identifiable service sources.	Class types break the class categories into further individual entities and suggest ways of measuring the associated ecosystem service output.	Possible indicators based on information potentially available within the BENTHIS project	
Provisioning	Nutrition	Biomass	Wild plants, algae and their outputs	*Plants, algae by amount, type*	Extent of specific habitat (e.g. seagrass), Biomass total or per specific taxa	Seaweed (e.g. Palmaria palmata = dulse, dillisk) for food
Provisioning	Nutrition	Biomass	Wild animals and their outputs	*Animals by amount, type*	Biomass total or per specific taxa	Marine fish (plaice, sea bass etc.) and shellfish (i.e. crustaceans, molluscs), Includes commercial and subsistence fishing for food
Provisioning	Materials	Biomass	Fibres and other materials from plants, algae and animals for direct use or processing	*Material by amount, type, use, media (land, soil, freshwater, marine)*	Biomass total or per specific taxa	Sponges and other products, which are not further processed; material for production e.g. chemicals extracted or synthesized from algae, plants and animals such as turpentine, rubber, flax, oil, wax, resin, soap (from bones), natural remedies and medicines (e.g. chondritin from sharks), dyes and colours, ambergris (from sperm whales used in perfumes); Includes consumptive ornamental uses.
Provisioning	Materials	Biomass	Materials from plants, algae and animals for agricultural use		Biomass total or per specific taxa	Plant, algae and animal material (e.g. grass) for fodder and fertilizer in agriculture and aquaculture;

SECTION	DIVISION	GROUP	CLASS	CLASS TYPE	INDICATOR	EXAMPLES
Provisioning	Materials	Biomass	Genetic materials from all biota		Taxonomic diversity	Genetic material (DNA) from wild plants, algae and animals for biochemical industrial and pharmaceutical processes e.g. medicines, fermentation, detoxification; bio-prospecting activities e.g. wild species used in breeding programmes etc.
Provisioning	Energy	Biomass-based energy sources	Plant-based resources	*By amount, type, source*	Biomass total	Wood fuel, straw, energy plants, crops and algae for burning and energy production
Regulation & Maintenance	Mediation of waste, toxics and other nuisances	Mediation by biota	Bio-remediation by micro-organisms, algae, plants, and animals	*By amount, type, use, media (land, soil, freshwater, marine)*	Biomass per Bioturbation mode trait	Bio-chemical detoxification/decomposition/mineraliztion in land/soil, freshwater and marine systems including sediments; decomposition/detoxification of waste and toxic materials e.g. wastewater cleaning, degrading oil spills by marine bacteria, (phyto)degradation, (rhizo)degradation etc.
Regulation & Maintenance	Mediation of waste, toxics and other nuisances	Mediation by biota	Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals	*By amount, type, use, media (land, soil, freshwater, marine)*	Biomass per apprpriate Feeding mode trait (e.g. Suspension & Filter-feeders)	Biological filtration/sequestration/storage/accumulation of pollutants in land/soil, freshwater and marine biota, adsorption and binding of heavy metals and organic compounds in biota
Regulation & Maintenance	Mediation of waste, toxics and other nuisances	Mediation by ecosystems	Filtration/sequestration/storage/accumulation by ecosystems	*By amount, type, use, media (land, soil, freshwater, marine)*	Proportion of specific habitat (ie. Soft sediment) disturbed, Amount of sediment suspended	Bio-physico-chemical filtration/sequestration/storage/accumulation of pollutants in land/soil, freshwater and marine ecosystems, including sediments; adsorption and binding of heavy metals and organic compounds in ecosystems (combination of biotic and abiotic factors)

SECTION	DIVISION	GROUP	CLASS	CLASS TYPE	INDICATOR	EXAMPLES
Regulation & Maintenance	Mediation of waste, toxics and other nuisances	Mediation by ecosystems	Dilution by atmosphere, freshwater and marine ecosystems		Proportion of specific habitat (ie. Soft sediment) not disturbed, Amount of sediment suspended	Bio-physico-chemical dilution of gases, fluids and solid waste, wastewater in atmosphere, lakes, rivers, sea and sediments
Regulation & Maintenance	Mediation of flows	Mass flows	Mass stabilization and control of erosion rates	*By reduction in risk, area protected*	Extent of specific habitat (e.g. seagrass)	Erosion / landslide / gravity flow protection; vegetation cover protecting/stabilizing terrestrial, coastal and marine ecosystems, coastal wetlands, dunes; vegetation on slopes also preventing avalanches (snow, rock), erosion protection of coasts and sediments by mangroves, seagrass, macroalgae, etc.
Regulation & Maintenance	Mediation of flows	Liquid flows	Flood protection	*By reduction in risk, area protected*	Extent of specific habitat (e.g. seagrass)	Flood protection by appropriate land coverage; coastal flood prevention by mangroves, seagrass, macroalgae, etc. (supplementary to coastal protection by wetlands, dunes)
Regulation & Maintenance	Maintenance of physical, chemical, biological conditions	Life cycle maintenance, habitat and gene pool protection	Maintaining nursery populations and habitats	*By amount and source*	Extent of specific habitat (e.g. seagrass, gravel)	Habitats for plant and animal nursery and reproduction e.g. seagrasses, microstructures of rivers etc.
Regulation & Maintenance	Maintenance of physical, chemical, biological conditions	Pest and disease control	Pest control	*By reduction in incidence, risk, area protected*	Taxonomic diversity	Pest and disease control including invasive alien species
Regulation & Maintenance	Maintenance of physical, chemical, biological conditions	Soil formation and composition	Weathering processes	*By amount/concentration and source*	Biomass per appropriate Bioturbation mode trait	Maintenance of biogeochemical conditions of soils including fertility, nutrient storage, or soil structure; includes biological, chemical, physical weathering and pedogenesis

Section	Division	Group	Class	Class type	Indicator	Examples
Regulation & Maintenance	Maintenance of physical, chemical, biological conditions	Soil formation and composition	Decomposition and fixing processes		Biomass per appropriate Bioturbation mode trait	Maintenance of biogeochemical conditions of soils by decomposition/mineraliztion of dead organic material, nitrification, denitrification etc.), N-fixing and other biogeochemical processes;
Regulation & Maintenance	Maintenance of physical, chemical, biological conditions	Water conditions	Chemical condition of salt waters		Biomass per appropriate Bioturbation mode trait	Maintenance / buffering of chemical composition of seawater column and sediment to ensure favourable living conditions for biota e.g. by denitrification, re-mobilization/re-mineraliztion of phosphorous, etc.
Regulation & Maintenance	Maintenance of physical, chemical, biological conditions	Atmospheric composition and climate regulation	Global climate regulation by reduction of greenhouse gas concentrations	*By amount, concentration or climatic parameter*		Global climate regulation by greenhouse gas/carbon sequestration by terrestrial ecosystems, water columns and sediments and their biota; transport of carbon into oceans (DOCs) etc.
Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Physical and experiential interactions	Experiential use of plants, animals and land-/seascapes in different environmental settings	*By visits/use data, plants, animals, ecosystem type*	Extent of specific habitats, Taxonomic diversity	In-situ whale and bird watching, snorkelling, diving etc.
Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Physical and experiential interactions	Physical use of land-/seascapes in different environmental settings		Extent of specific habitats, Taxonomic diversity	Walking, hiking, climbing, boating, leisure fishing (angling) and leisure hunting

Section	Division	Group	Class	Class type	Indicator	Examples
Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Intellectual and representative interactions	Scientific	*By use/citation, plants, animals, ecosystem type*	Extent of specific habitats, Taxonomic diversity	Subject matter for research both on location and via other media
Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Intellectual and representative interactions	Educational		Extent of specific habitats, Taxonomic diversity	Subject matter of education both on location and via other media
Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Intellectual and representative interactions	Heritage, cultural		Extent of specific habitats, Taxonomic diversity	Historic records, cultural heritage e.g. preserved in water bodies and soils
Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Intellectual and representative interactions	Entertainment		Extent of specific habitats, Taxonomic diversity	Ex-situ viewing/experience of natural world through different media

SECTION	DIVISION	GROUP	CLASS	CLASS TYPE	INDICATOR	EXAMPLES
Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Intellectual and representative interactions	Aesthetic		Extent of specific habitats, Taxonomic diversity	Sense of place, artistic representations of nature
Cultural	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Spiritual and/or emblematic	Symbolic	*By use, plants, animals, ecosystem type*	Extent of specific habitats, Taxonomic diversity	Emblematic plants and animals e.g. national symbols such as American eagle, British rose, Welsh daffodil
Cultural	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Spiritual and/or emblematic	Sacred and/or religious		Extent of specific habitats, Taxonomic diversity	Spiritual, ritual identity e.g. 'dream paths' of native Australians, holy places; sacred plants and animals and their parts

SECTION	DIVISION	GROUP	CLASS	CLASS TYPE	INDICATOR	EXAMPLES
Cultural	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Other cultural outputs	Existence	*By plants, animals, feature/ecosystem type or component*	Extent of specific habitats, Taxonomic diversity	Enjoyment provided by wild species, wilderness, ecosystems, land-/seascapes
Cultural	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Other cultural outputs	Bequest		Extent of specific habitats, Taxonomic diversity	Willingness to preserve plants, animals, ecoystems, land-/seascapes for the experience and use of future generations; moral/ethical perspective or belief

6.2.4.4 Selection of operational indicators: a Norwegian example

Although the MSFD is not implemented in Norway, the Norwegian management plan contains some benthic indicators, covering several aspects of the seabed (van der Meeren and Pettersen, 2012; Anon., 2014) that can be aligned to the requirements of the MSFD (Table 6.8). Some of these indicators are operational, like the monitoring of the abundance and recruitment of the deep-sea shrimp (*Pandalus borealis*), soft-bottom biodiversity, biomass and species composition in bottom-trawl surveys, level of pollutants in sediments and benthic biota and abundance and distribution of the alien red king crab (*Paralithodes camtschaticus*). Métiers of the bottom-trawl activities, live tissues on corals and sponges and trends in red listed species are not yet operational indicators.

Table 6.8. Proposed operational indicators for the relevant aspects of state of the seafloor in the Norwegian management plans (Anon., 2009; 2011; 2013).

STATE CATEGORY	NORWEGIAN MANAGEMENT PLAN CRITERIA	NORWEGIAN INDICATORS	PROPOSED OPERATIONAL INDICATOR
Physical habitat	Physical damage, having regard to substrata characteristics	Type, abundance, biomass and areal extent of relevant biogenic substrata (6.1.1)	
		Extent of the seabed significantly affected by human activities for the different substratum types (6.1.2)	-Level of bottom-trawl activity, yet to be defined (North Sea and Skagerrak) -Level of pollution in sediments
Associated benthic community	Condition of benthic community	Presence of particularly sensitive and/or tolerant species (6.2.1)	-Live tissue cover on deep-sea corals and sponges -Trends in red listed species occurrence -Species composition and amount of benthos in survey trawl hauls -Level of selected pollutants in deep-sea shrimps and blue mussels
		Multimetric indices assessing benthic community condition and functionality, such as species diversity and richness, proportion of opportunistic to sensitive species (6.2.2)	-Biodiversity in grab samples from soft bottoms -Species composition and amount of benthos in survey-trawls hauls (Barents Sea)
		Proportion of biomass or number of individuals above some specified length/size (6.2.3) Parameters describing the characteristics (shape, slope and intercept) of the size spectrum of the benthic community (6.2.4)	-Biomass of trawled deep-sea shrimp surveys (Skagerrak, North Sea) -Assessment of 1y recruits in trawled surveys for deep-sea shrimps -Abundance of red king crabs /alien species) (Barents Sea) -Live tissue cover on deep-sea corals and sponges

6.2.5 Ghost fishing

As a part of a national project, DTU Aqua is planning to write a concise review in autumn 2014 of the documented physical impacts of bottom-set gillnets on benthic flora and epifauna, focusing mainly on temperate seas. This review will also include general aspects relating to ghost fishing by derelict nets (i.e. not traps, creels, etc.). A preliminary outline of the review is as follows:

- Physical impacts of set gillnets (Subdivided into fishing and set-ting/hauling):
  - Physical impacts of bottom-set gillnets on marine flora, epifauna and benthic habitats;
  - Examples (if any) of fisheries technological approaches to mitigate physical impacts of bottom-set gillnets.
- Ghost fishing:
  - Reasons for loss of bottom-set gillnets;
  - Review of quantification/estimates of loss of nets;
  - Documented and estimated catches of fish and benthos (e.g. crabs) in derelict nets;
  - Main factors affecting perseverance of the ability of derelict gears to fish;
  - Technological and management initiatives to mitigate loss of nets and/or facilitate their retrieval;
  - Ghost nets as marine macrolitter and/or source of microliter.

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7   ToR e) Recommend priority areas of study to determine the ecosystem consequences of landing obligations/discard bans, including survival associated with releasing fish caught

7.1   Background

One of the objectives in the Reform of the Common Fisheries Policy (CFP) is the avoidance and minimization of unwanted catches. This is envisaged in the "landing obligation", which regulates that all catches of species which are subject to catch limits shall be retained on-board fishing vessels and landed (EU, 2013).

Another objective mentioned in this EU regulation, Article 2 (EU, 2013) states that:

> "The CFP shall implement the ecosystem-based approach to fisheries management so as to ensure that negative impacts of fishing activities on the marine ecosystem are minimized, and shall endeavour to ensure that aquaculture and fisheries activities avoid the degradation of the marine environment."

The "ecosystem-based approach to fisheries management" was further defined in Article 4(9) as:

> "an integrated approach to managing fisheries within ecologically meaningful boundaries which seeks to manage the use of natural resources, taking account of fishing and other human activities, while preserving both the biological wealth and the biological processes necessary to safeguard the composition, structure and functioning of the habitats of the ecosystem affected, by taking into account the knowledge and uncertainties regarding biotic, abiotic and human components of ecosystems."

Other international legislative documents also indicate the requirement of European countries to adopt the ecosystem-based approach to fisheries management, principally the Marine Strategy Framework Directive (EC, 2008; Jennings and Rice, 2011).

While the landing obligation and the Reform of the Common Fisheries Policy seek to ensure that fishing practices do not harm the ability of fish populations to reproduce by fishing more selectively and phasing out the practice of discarding unwanted fish, the ecosystem consequences of landing catches so far discarded were not addressed. Reductions in discarding practices may not always be beneficial but could potentially cause some unexpected effect on the ecosystem, at least in the short term (Zhou, 2008). One example of such unintended consequence is the change in the subsidies provided by discarded materials to various components of the foodweb. Scientists should prevent discard management becoming an example of 'faith-based fisheries' (Hilborn, 2006).

WGECO adopted therefore the following Term of Reference:

"Recommend priority areas of study to determine the ecosystem consequences of landing obligations/discard bans, including survival associated with releasing fish caught".

7.2 The current knowledge base

7.2.1 Current discarded amount and composition

The part of the catch which is not retained on board fishing vessels is defined as the discarded fraction of the catch. Within the Reform of the Common Fisheries Policies the lack of reduction of discards has been put forward as one of the reasons for limited stock recovery and as a waste of marine resources(e.g. Catchpole *et al.*, 2005; Condie *et al.*, 2013; Depestele *et al.*, 2011; Morandeau *et al.*, 2014; Uhlmann *et al.*, 2013). Articles 14 and 15 ("Landing obligation") of the EU Regulation (EU, 2013) intend to curtail discarding by obliging fishermen to land the total catch of all species that have catch quota limitations, or minimum landing sizes in the Mediterranean. Estimates of discards have shown high and variable levels. The North Sea fisheries may be responsible for the highest level of discards in the world (Kelleher, 2005), although discard rates have dropped in recent years (Enever *et al.*, 2009). Historical estimates of total discards were up to 789 000 tonnes in the North Sea, where discards included all living animals (from benthic invertebrates, non-commercial fish to quota-regulated species). This represented 4% of total biomass of fish and 22% of total landings (Garthe *et al.*, 1996).

Here we present recent discard estimates based on the STECF database, based on data from the DCF data call. Details on sampling and analysis can be found in the report produced by the STECF Expert Working Group (STECF, 2013a). We only present the 2012 data, as more details can be found in their report. Our intention is an indication of the orders of magnitude of discarding (in biomass) to appraise the amount of potential decrease in subsidies to the ecosystem as a consequence of eliminating discards from species under the landing obligation, and to highlight the issues for ecosystem assessments. Figure 7.1 summarizes the results and illustrates that most of the reported discards logically fall in the TAC species category, except for the Baltic Sea, the North Sea and the Bay of Biscay. The high discards in the Baltic Sea are due to high discards of Pleuronectiformes, which could both be TAC (e.g. European plaice) or non-TAC species (e.g. European flounder). In the North Sea the discards of Brown shrimp (*Crangon crangon*) should be taken into account. The reported discards in the Bay of Biscay exclusively account for Belgian beam trawlers and hence present an important underestimation of discarding in this region (see below). The STECF discard estimates were considered the best available European-wide database for discards. However, these data are estimates based on a number of simplifications, such as the procedure used to estimate missing data from unsampled combinations of area, métier, and/or season. The index of Discard Coverage (DQI) denotes the landings for which discards are sampled (Ld, tonnes) and compare them to the overall landings (L, tonnes):

$$DQI = \Sigma\, Ld\, /\, \Sigma\, L$$

DQI is expressed by stock, fishery and Member State as the proportion of national landings covered by discard estimates in relation to the total national landings. Discard estimates are provided for <60% of the total landed biomass in all regions, except for the fully documented fisheries (Figure 7.2).
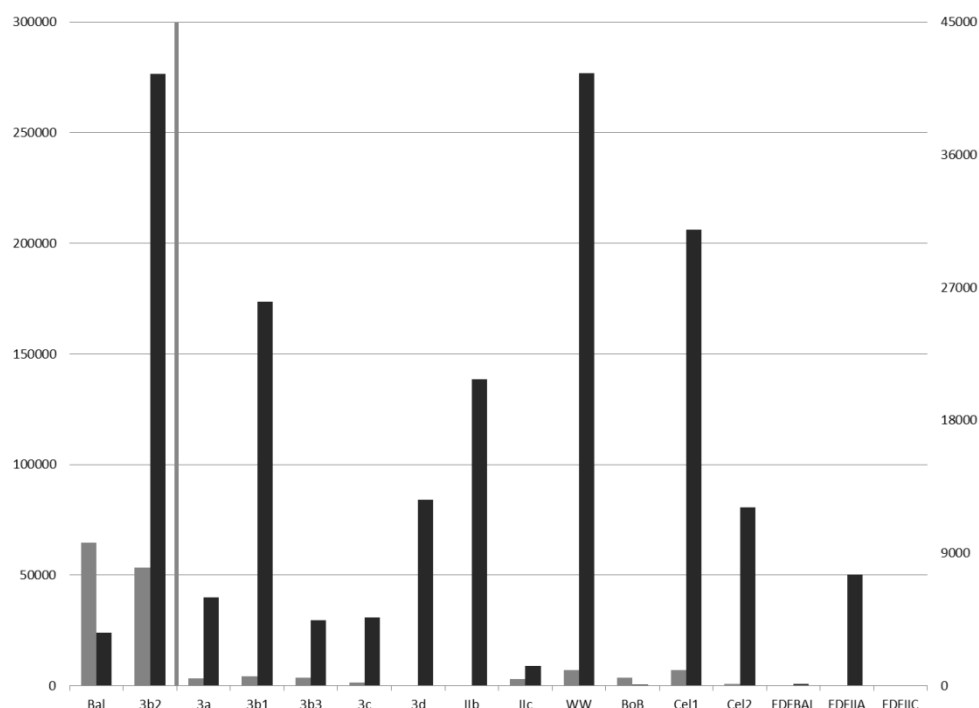
Figure 7.1. Total reported discards (in tonnes) in European waters in 2012, based on STECF data (STECF, 2013a). The left (light grey) bars indicate discards from species which could not be attributed to any TAC species. Dark grey bars on the right side include discards of TAC species. Note that discards identified as Pleuronectiformes are included in the category of non-TAC species, because it was unclear whether these are flounder (*Plathythys flesus*) or other species such as European plaice (*Pleuronectes platessa*). Discards of TAC-species are hence a minimum estimate from the STECF data. Bal: Baltic Sea, 3b2: North Sea, 3a: Kattegat, 3b1: Eastern English Channel, 3c: Irish Sea, 3d: waters west of Scotland, IIb: ICES Division VIIIc and IXa, IIc: Eastern English Channel, WW: Western waters, BoB: Bay of Biscay, Cel1: Celtic Sea (VIb,c,e,f,g,h,j,k), Cel2: Celtic Sea (VIIfg), FDF: fully documented fishery. Details can be found in STECF, 2013a, Appendix 2-2: http://stecf.jrc.ec.europa.eu/web/stecf/ewg1313.

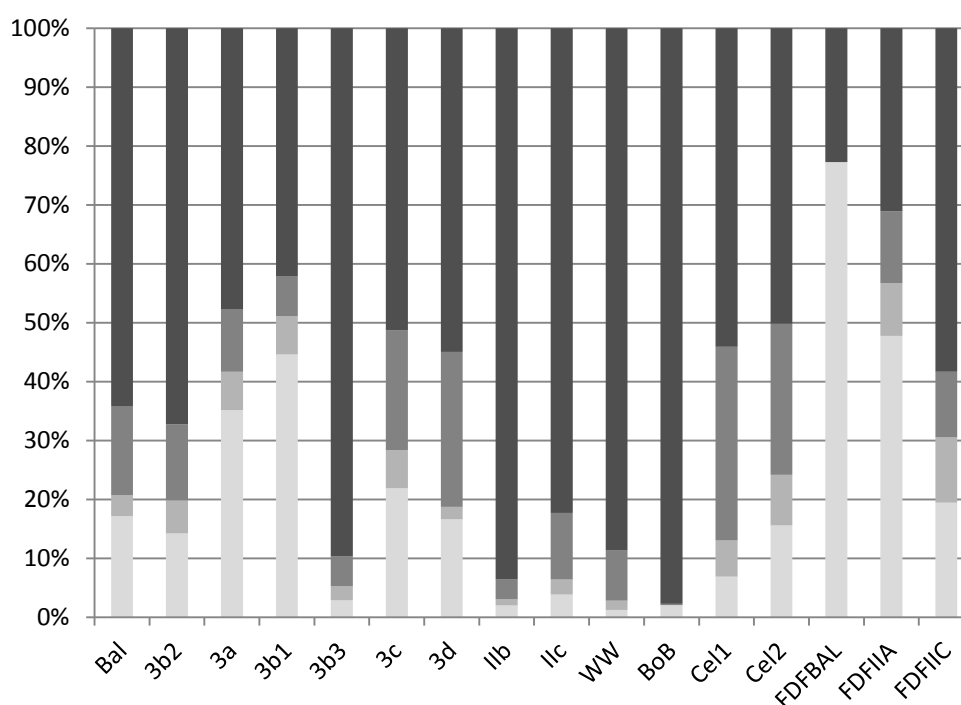**Figure 7.2. The percentage of each area (see Figure 7.1 for abbreviations) for which the discards are sampled in relation to all landed biomass. Percentages are based on area-based fisheries, defined by stock/fish species and fishing gear. Some fishing gears are not sampled in certain areas (e.g. upper, dark grey), while others have been sampled only for a limited number of landed species. Greyscale relates to the DQI: from light to dark grey >67%, 34–66%, 1–33% and 0%.**

The fate of discards partially depends on the feeding behaviour of seabirds. Since they are the first to encounter the discarded fraction of the catch, their selection process is considerably important. The morphology and size of discarded organisms determine to a large extent whether discards can be swallowed by seabirds. They can be classified in the following food preference groups: roundfish, flatfish, elasmobranchs, cephalopods and benthic invertebrates (Camphuysen *et al.*, 1995; Garthe *et al.*, 1996; Xavier *et al.*, 2013). The reported STECF discards are dominated by roundfish species, except for the North Sea (flatfish dominated), Kattegat (roundfish, flatfish and invertebrates), the Irish Sea and the fully documented fisheries (Figure 7.3).
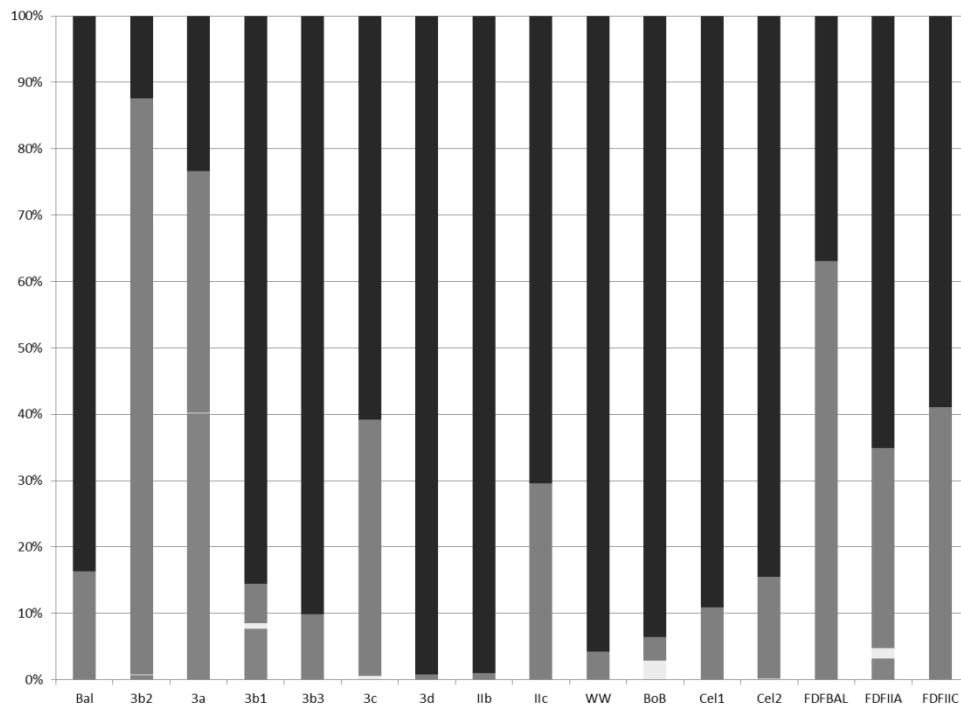
Figure 7.3. Attribution of STECF reported discards to discard categories that are consumed by seabirds with different capture efficiencies due to their morphology: roundfish, flatfish, elasmobranchs and benthic invertebrates (from dark to light grey).

Ecosystem effects of discards do not discriminate between non-TAC species and non-commercial species *sensu lato*. However, area-specific estimates of total discards are lacking, hampering an estimation of the magnitude of the relative change in discard biomass due to the landing obligation. Discards in the French fisheries are fully quantified (Cornou *et al.*, 2013). Although limited to the fisheries of one Member State, the importance of discards from quota species can be compared with those from non-quota species, including non-commercial fish and benthic invertebrate species (Figure 7.4). These figures indicate that discards in the Bay of Biscay for instance from the STECF figures underestimate total discards, even if only French fisheries are accounted for. Second, the importance of discards that do not fall under the landing obligation was illustrated. The importance of discards from non-quota species is comparable for most areas except for the Eastern English Channel and the North Sea, where quota-discards are higher. These figures highlight the importance of accounting for all species in discard data to assess ecosystem effects.
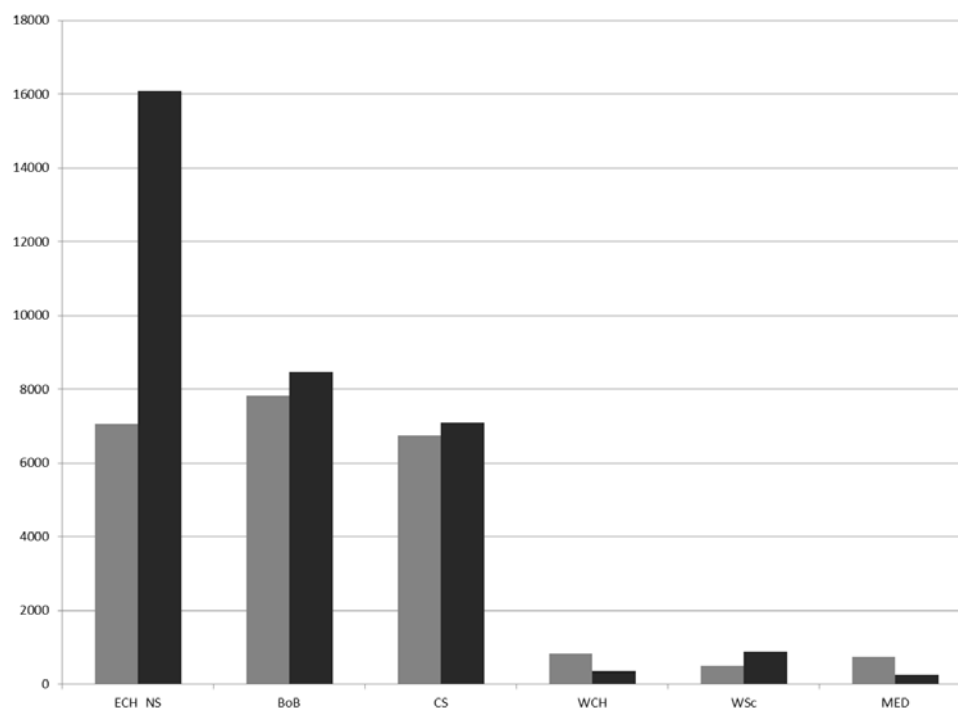
Figure 7.4. Discards of French fisheries in six areas: (1) Eastern English Channel (ECH_NS), (2) Bay of Biscay (BoB), Celtic Seas (CS), Western English Channel (WCH), Waters west of Scotland (WSc) and the Mediterranean (MED) (modified from Cornou *et al.*, 2013). The left (grey) bars indicate discards from species which could not be attributed to any TAC species. Black bars on the right side include discards of TAC species.

7.2.2 Discard survival

The ecosystem consequences for fish stocks and the marine ecosystem will also depend on the implementation of the EU Regulation (EU, 2013). The implementation of the landing obligation can be interpreted in several ways in relation to the exemption possibilities related *inter alia* to the demonstration of high survival rates in Article 15, paragraph 2(b). We refer to other fora for further details on this implementation, e.g. STECF (2013b) and the ICES Workshop on Methods for Estimating Discard Survival (ICES WKMEDS). The consequences of the decision on whether a species will be allowed for an exemption of the landing obligation will undoubtedly have consequences for the fish stocks, either positive or negative. High discards of species with a low survival rate can for instance influence its stock biomass, but currently there are no estimates available hereof. Both estimates of survival and its consequences for the stock as well as the role of the survivors and/or the reduced fitness of discarded individuals are poorly understood. Therefore WGECO re-iterate the recommendations of ICES WKMEDS with respect to estimation of the survival components most relevant to ecosystem considerations, namely:

- discard survival rate, including predation effects, for particular conditions;
- discard survival rate, including predation effects, representative of the management unit.

7.2.3 Ecosystem use of discards

Fisheries generate carrion as a result of material discarded. As discarding has been ongoing for decades, communities may have developed that are reliant on discards as

a food source (Kaiser and Hiddink, 2007). In general, the fate of discarded organisms has only been the topic of limited investigations (Wassenberg and Hill, 1990) (Figure 7.5) and it is e.g. unclear whether the increases in the population sizes in scavenging seabirds partially attributed to discarding practices (Votier *et al.*, 2004) might be mirrored in changes in populations of meso-pelagic and benthic scavengers.
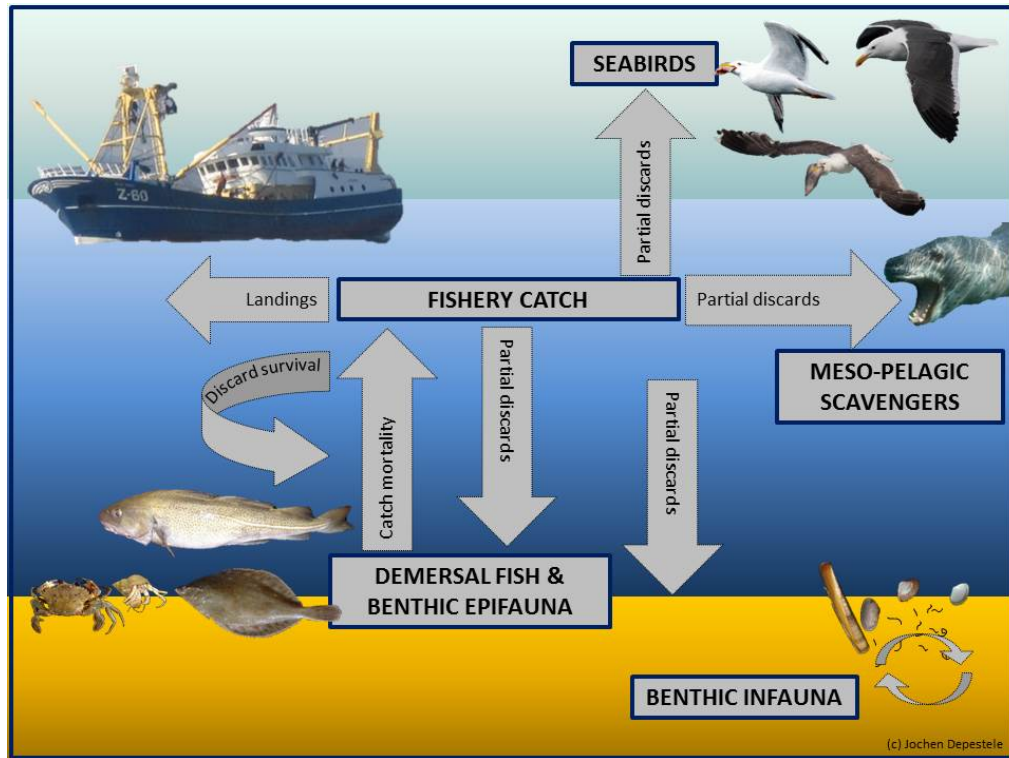

Figure7.5. Schematic representation of the endpoints of fishery catches. Discarded organisms can be landed, or discarded. The fate of discards can be the consumption by scavenging seabirds and meso-pelagic scavengers. When they are not consumed, they reach the seafloor at the advantage of benthic scavengers or they survive the capture-and-discard process and return to the fish or invertebrate community.

Both the positive and negative effects on seabirds have been investigated more than effects on any other ecosystem component (Bicknell *et al.*, 2013; Votier *et al.*, 2013). At least 143 seabird species worldwide (52% of the global taxonomic diversity) make use of discards to some extent (Oro *et al.*, 2013). Discards have historically shaped many aspects of seabird foraging, distribution and population dynamics. EU seabird scavengers are dominated by a relatively small number of large generalist taxa. Various seabird species use discards and offal as trophic resources, and some species are believed to have increased in numbers as a result of a greater availability of food via discards (Martinez-Abrain *et al.*, 2002). They forage on both live prey and fishery wastes, eventually favouring the latter when the former becomes scarce. The most exhaustive estimate of discard consumption was given for the North Sea in the 1990s. The total amount of fishery waste (without offal, defined as organic material from gutting fish) in the North Sea was estimated at 726 200 tonnes of which seabirds consumed 255 000 tonnes. About 150 000 tonnes of invertebrates were discarded, but hardly consumed by seabirds (~9000 tonnes). In total, discards potentially supported up to 5.9 million seabirds (Garthe *et al.*, 1996). However, given the uncertainties in the discard estimates, caution is needed (Stratoudakis, 1999).

In studies of the composition and fate of catch and bycatch in a *Nephrops* fishery in the Farne Deep in the North Sea, Evans *et al*. (1994) observed that 88% of the catch was made up of bycatch consisting of mostly unmarketable fish (34 species) and 23 invertebrate taxa, including *Nephrops*. The authors estimate that >70% of discard was consumed by seabirds near the surface. Catchpole *et al*. (2006) studied whether or not discarded material from the English *Nephrops* fishery has a positive effect on marine scavenger populations and found that seabirds utilized an estimated 57% of the discarded material.

What happens to discards that are not consumed by seabirds is less well understood. Some discards might survive (see above), or be scavenged upon in the water column (Pon *et al*., 2012). Drazen *et al*. (2012) report on the temporal changes in grenadier's abundance as a consequence of changes in abundance of carrion supply from surface living nekton. Their results suggest that some abyssal fishes' population dynamics are controlled by the flux of large particles of carrion. The effects of discards-generated carrion however are not known, but might be of importance. Hill and Wassenberg (1990; 2000) indicated that a third to half of the discarded fish and cephalopods from prawn trawlers in Australia might float when discarded, and be scavenged upon by birds and pelagic organisms such as dolphins and sharks. The remaining fraction, including crustaceans, sunk to the seafloor where it was scavenged by epibenthic invertebrates.

Several studies have been carried out to determine the fate of discarded organisms and the effects of discarding on benthic scavengers. A number of these studies have focused on *Nephrops norwegicus* fisheries (Bergman *et al.,* 2002; Evans *et al.,* 1994; Catchpole *et al.,* 2006) and beam trawl fisheries in the North Sea (Groenewold and Fonds, 2000), likely due to the high discard rates in these fisheries (Scheveningen Group, 2014). Using bait piles with a composition typical of discards from Clyde Sea *Nephrops* trawling, Bergmann *et al*. (2002) focus on the fate of discarded material from and identification of the scavengers attracted to the invertebrates discarded. The authors were not able to quantify the extent to which discards subsidize benthic communities. Using a combination of field studies and a bioenergetic model Catchpole *et al*. (2006) studied whether or not discarded material from the English *Nephrops* fishery has a positive effect on marine scavenger populations. In their study most discarding (83%) took place over the fishing grounds. Bioenergetic model estimates indicate that the energy input from discards in this fishery potentially could provide the identified scavengers with 37% of their energetic requirements locally during the fishing season. Fulton *et al*. (2005) also predicted an increase in those groups (e.g. deposit-feeders) that scavenge carrion (or detritus), or benefit from discarding in other ways.

In the Southern North Sea, Groenewold and Fonds (2000) studied beam trawl fisheries and the effects of discards and damaged benthos on benthic scavengers by deploying 370 baited traps with different kinds of representative carrion species in 14 locations. Modelling studies indicate that, in contrast to the Catchpole *et al*. (2002) study, the direct importance of discard as additional food for scavengers is relatively small. According to the authors, however, the importance may be relatively larger for scavenging fish than for invertebrates. The authors conclude that beam trawling may lead to shortcuts in trophic relationships, i.e. enhancing secondary production. These findings are confirmed by Kaiser and Hiddink (2007). Their study balances the decreases in benthic biomass from fishing and the short-term production generated by carrion. The production of carrion compensates for 22% of the reduction in biomass production from direct mortalities.

The decomposition of discard at the sediment surface can lead to alteration of nematode communities in the sediment (Lenihan and Peterson, 1998; Franco *et al.*, 2008) reflecting biogeochemical changes in the sediment. Decomposition is associated with oxygen stress underlying the dead organic material (Trush, 1986), which in turn will affect important mineralization processes in the sediment (Aller, 1988). These benthic mineralization processes are very important for the marine ecosystem as a whole, as they provide the water column with nutrients needed for primary production on the one hand, and counteract eutrophication by the removal of nitrogen as N_2 through denitrification processes. However, we are currently unaware of knowledge on the effect of discards on infauna functioning.

The combined list of (mainly North Sea) scavengers cited above hence include hermit crab *Pagurus bernhardus*, swimming crab *Liocarcinus depurator*, whelks, *Nephrops*, common crabs *Carcinus maenas*, edible crab *Cancer pagurus*, hagfish *Myxine glutinosa*, ophiurids, small gadoids, amphipods (*Orchomene nanus, Scopelocheirushopei*), grenadier, dab *Limanda*, plaice *Pleuronectes platessa*, whiting *Merlangius merlangus*, dragonet *Callionymus lyra*, grey gurnard *Eutrigla gurnardus*, doplhins and elasmobranchs.

7.3 Effects of a landing obligation

7.3.1 Changes in fishing gears

The fishing pattern and fisheries' catches are expected to change considerably as a consequence of the landing obligation, both through changes in gear selectivity, fishing behaviour (e.g. skippers' skills, vessel characteristics, etc.) as well as changes in spatial and temporal distribution of fisheries. A reduction of catches through increased gear (size and species) selectivity implies that the selection process will take place underwater rather than on-board the fishing vessel. This may imply that a larger part of mortality may be unaccounted for, as i.e. the relative contribution of escape mortality vs. discard mortality will change. WGECO recommend that these unknown, potential sources of unaccounted mortality be investigated.

7.3.2 Changes in fishing patterns

The potential effects of a discard ban on the distribution of the fishing activities was explored by Batsleer *et al*. (2013) using a model of the French mixed fisheries in the Eastern English Channel. They examined in particular the performance of two different management scenarios: (1) individual quota management with a tolerance for discarding and (2) individual quota management in combination with a discard ban, using a dynamic state variable model. The model evaluates a time-series of decisions taken by fishers to maximize profits within management constraints. Compliance to management was tested by applying an in-height varying fine for exceeding the quota. Then they evaluated the consequences of individual cod quota in both scenarios with respect to over-quota discarding, spatial and temporal effort allocation and switching between métiers. Individual quota management without a discard ban hardly influenced fishers' behaviour as they could fully utilize cod quota and continue fishing other species while discarding cod. In contrast, a discard ban forced fishers to reallocate effort to areas and weeks in which cod catch is low, at the expense of lower revenue. This modelling illustrates the potential influences of redistribution of fisheries as a consequence of quota restrictions. The presence of "choke" species is expected to have an important influence (Baudron and Fernandes, 2014).

7.3.3   Exemptions and exceptions

The actual consequences of the landing obligation will depend to a large extent on the way Article 15 will be implemented in each marine region. Member states and Regional Advisory Councils are commissioned to elaborate regional discard management plans before the landing obligation comes into force for the various fisheries. These discard management plans will, among other, list the species and stocks benefiting from exemptions or exceptions to the landing obligation in each fishery. For example, species with "high" survival might be exempted from the landing obligation; the ecosystem consequences of the new regulation will then depend on the residual mortality rate after discarding (see Section 7.2.2 above).

Another exception to the landing obligation will be granted to the catch used as live bait. Therefore baited fisheries, either by traps or longlines, may reduce the impact of the landing obligation. Thus monitoring areas with extensive fisheries with longlines or traps may provide insight about the consequences of the new regulation.

7.3.4   Possible effects on the foodweb of the landing obligation

As a wide diversity of organisms exploits fishery discards, ranging from aerial scavengers to seafloor inhabitants, changes in carrion production could lead to substantial ecosystem consequences (Beasley *et al.*, 2012; Lewison *et al.*, 2012; Votier *et al.*, 2010). With the introduction of the landing obligation, the discarded amount of commercial fish and invertebrates is expected to decrease considerably. A direct consequence of banning discards is therefore the creation of a food shortage for scavenging species. The effect of this shortage depends on the ability of the scavengers to compensate by switching to other food sources. This may limit the direct effects on these species, but may cause cascading effects on other species through increased predation or competition.

7.3.4.1 Seabirds

Some seabird species feeding on discards can potentially shift to feeding on other prey. This sometimes entails increased predation on other bird species with resulting negative effects on their populations. Some species have the potential to move into habitats that have not reached their carrying capacities, e.g. herring gulls in coastal cities. However, in general, the reduction in food might be expected to lead to decreased populations of the species most dependent on discards such as large generalist seabird species. Bicknell *et al.* (2013) further highlights the knowledge gaps in research on seabirds in relation to discards. These include a lack of understanding of the consequences in the non-breeding season, the effects on immature birds and the implications of feeding on a food source with low nutritional value ("junk food hypothesis"). The potential of changing foodweb dynamics in the ecosystem were also highlighted.

7.3.4.2 Other scavengers

The species identified to feed on discards range from marine mammals over commercial and non-commercial fish to benthos. The information currently available indicates that there may be local effects of the lower discard rates, but the knowledge is insufficient to determine larger scale effects. There are several aspects that are poorly described in the literature:

- The relative contribution of discards to the food consumption of the large-scale scavenger community is unknown as is the conversion efficiency of the ingested matter to scavenger biomass;
- The extent to which the scavenging community will be able to compensate by feeding on alternative prey is unclear. Depending on this, the further unknowns become:
  - If full compensation is attained, what is the effect on their new prey?
  - If full compensation is not attained, what is the effect of the potential reduction in scavengers on the mortality of their alternative prey?

As a general observation, species specialising on scavenging or species with low mobility will be more likely to exhibit decreases in local abundance and/or biomass than other species.

7.3.4.3 Indirect effects

Species with a generalist diet or a high mobility will tend to increase their predation on other food sources, hence leading to indirect effects on alternative prey. In addition, changes in the distribution and selectivity of the fishery modify ecosystem impacts. Given that the extent and direction of changes in the fishery are unknown at present and that the knowledge of the potential prey substitution is very limited, the expected effects of a landings obligation on the ecosystem can only be inferred in very broad terms.

Increased competition for food

Changes in abundances of scavenger populations have the potential to alter community dynamics through changes in competition and resource availability of other available food. Generalist scavengers may switch to other prey and hence increase competition for these prey sources whereas specialist predators may decrease in abundance and hence decrease the mortality on secondary prey sources. The degree of competition and predatory release is likely to change over time as the system adapts to a state with limited discards. Further knowledge is needed about the abundance of two groups of scavengers to determine the strength of each of the two responses.

Decreased fishing mortality of small fish

The landing obligation is often thought to enhance selection for larger fish and decrease fishing pressure on choke species. All else equal, this is likely to decrease fishing mortality, thus to potentially increase the abundance of small to medium sized fish, as well as the abundance of fish with current low quotas. An increase in small fish may enhance productivity of non-scavenging seabirds, which have been reported to be food limited in certain cases (Rindorf *et al.,* 2000; Frederiksen *et al.,* 2004; Engelhard *et al.,* 2014). In contrast, reports of food limitation in marine mammals and predatory fish are rarer and generally restricted to areas with a low diversity of prey (Steinarsson and Stefánsson, 1996; Mello and Rose, 2005; Eero *et al.,* 2012). Hence, areas which currently have a high abundance and diversity of prey, such as the North Sea, may not experience changes in predatory fish growth but may exhibit a greater mortality of large specimens.

Changes in energy conversion efficiency

Energy conversion efficiency from primary production to scavenger biomass changes when a predator feeds directly on naturally available prey rather than on prey species that has undergone the capture-and-discarding process. To this end, the balance between naturally available prey and the short-cut delivery of food through discarding needs to be investigated. Reducing the number and length of pathways linking scavenging fish to their prey simplifies foodwebs, and is arguably resulting in a decreasing ability of predators to switch between prey. This was suggested to make them more vulnerable to prey fluctuations (Pauly *et al.*, 2002; Wilson and Wolkovich, 2010).

7.3.5 Adequate monitoring

EU regulation stresses that the best available scientific advice requires harmonized, reliable and accurate datasets (EU, 2013). Biological data collection particularly requires information on catches, including discards and survey information on fish stocks. Besides this, the EU regulation states that (Article 25 in EU 2013) the assessment should be enabled of (1) exploited marine biological resources, as well as (2).

> "The level of fishing and the impact that fishing activities have on the marine biological resources and on the marine ecosystems".

Elucidating the ecosystem effects of discarding is embraced in the latter. This requires that data are collected in such a way that contrasts between discarding and no discarding can be analysed in a sound scientific manner. The priority areas highlighted above need different types of information, related to (1) catch monitoring, (2) ecosystem surveys, including scavengers and (3) fishing distribution.

7.3.5.1 Catch monitoring

The Reform of the Common Fishery Policy proposes to collect detailed and accurate documentation of all fishing trips (Article 15 (13) of the EU regulation, EU 2013), but does not specify which data this implies. The monitoring of the marketable fraction includes that information is collected on the total catch of commercial species that fall within catch limitations. This includes comprehensive, complete and reliable documentation of both the landings and the discards, which may be achieved by fully documented fisheries. The traditional sources of scientific information can be used such as market sampling, observer trips, catch and logbook returns. Trip and haul-based information can be collected such as length and age-based data for the commercial species under catch limitations. Several new systems are present to handle issues of the traditional mechanisms of data collection, such as fleet coverage and bias in discard observer data originating in behavioural changes when observers are onboard fishing vessels (Benoît and Allard, 2009). While remote electronic monitoring and/or self-sampling can improve the coverage of data collection, they are less suitable to collect data on species which are not landed. Closed-circuit television (CCTV) cameras (Kindt-Larsen *et al.*, 2011) are not developed for instance to collect the abundance of non-commercial species, such as the high percentage of discarded epibenthic invertebrates in beam trawl fisheries. An overview of the strengths, weaknesses, opportunities and threats for each of the technologies and approaches has been given in Mangi *et al.* (2013). It is key for the assessment of ecosystem consequences of the landing obligation that data on non-commercial species and other bycatch continue to be collected.

7.3.5.2 Ecosystem surveys, including scavengers

Understanding the consequences of the landing obligation on other (non-fish) ecosystem components requires monitoring of changes in abundance and distribution of all scavenger species or a suite of indicator species that represent a range of ecosystem components (e.g. seabirds, meso-pelagic organisms, benthic invertebrates, etc.). Information on scavenging seabirds for instance could be obtained from the European Seabirds At Sea (ESAS), while epibenthic data can be collected from trawl surveys (e.g. IBTS & BTS). However, this would probably require additional monitoring on these surveys, as the GOV, and even the large beam trawl used on BTS are probably not optimal epibenthic samplers (see Section 9.2.1). To enable the identification of consequences of the changes in discarding, it is important to establish a comparison between the current, baseline information and the altered situation after the implementation of the discard ban. The identification of locations with low and high discard rates and/or fishing effort is therefore needed. For further details on the monitoring design and the type of data to be collected, we refer to other chapters in this report (see Section 9.2.1).

7.3.5.3 Fishing distribution

The spatial and temporal fisheries distribution is expected to change due to the landing obligation. Changes in fishing location and period will lead to alterations of affected ecosystem components. Fisheries distribution should therefore be monitored by for instance the Vessel Monitoring System (VMS) for vessels larger than 12 m, and/or other means for smaller vessels (<12 m).

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8   ToR f) Review knowledge of the consequences to stocks of prey fish (and other parts of the ecosystem) of restoring/ maintaining stocks of predatory fish to MSY and recommend priority areas for study

8.1   Background and motivation for this ToR

The background to this ToR lays specifically in the possible issues arising for seabird and other dependent predator populations and breeding success from a combination of a landing obligation and MSY fishing targets for predatory fish. Seabirds depend heavily on forage fish species such as sandeel. Forage fish are important as prey for many predatory fish, and if these stocks increase, this has been suggested as a cause of forage fish decline based on both ecosystem modelling and analyses of dataseries. Seabirds are also known to make use of discards, which are expected to diminish following the landing obligation. Thus the possibility exists that seabirds in particular may face two threats to their food supply almost simultaneously. While this ToR was inspired by this particular case, the issue probably has relevance for many predator–prey interactions as commercial stocks move towards MSY targets. Under this ToR, we therefore review the evidence to support the generality of suggested effects of rebuilding predator stocks and following this review, recommend priority areas for future research.

8.1.1   Fishing at MSY

Fishing under the current MSY approach implies that all species will be fished at their respective F_{MSY} rates. These are set based on the current weight-at-age, selection pattern, natural mortality and stock–recruitment relationship and will be updated as these parameters change over time. Within this framework, the yield of individual stocks may increase from current levels for predatory fish with low current biomass, whereas prey fish may experience increased natural mortality and lower yields. Realized biomass levels will depend on the rates of change and hence on the abundances of interacting species. Failing to adapt the F_{MSY} rates for species experiencing increased mortality is likely to lead to unprecautionary fishing. The effect of the biomass and hence yield of predatory fish on the yield of prey fish leads to expected trade-offs between the two. The further away the community is from MSY levels, the less predictable the outcome, as the models predict into regions where data support is limited.

8.1.2   Historical context of this question

Since the late 1980s, there have been several analyses of the potential effects of reducing fishing pressure on the demersal predator species, leading to increased stock abundance, and hence predation mortality. The Multispecies Assessment Working Group used MSFOR to project the consequences of increasing the mesh size in the roundfish (cod, haddock, and whiting) and saithe fisheries (Pope, 1991). The multispecies model projected that increased abundance of large predators would decrease the landings of seven of the nine species included in the model, including the prey species herring, sprat, Norway pout, and sandeel (Figure 8.1).
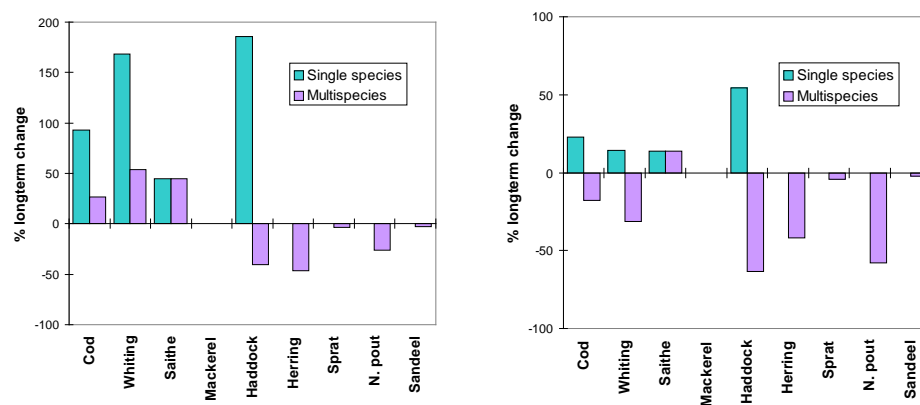
Figure 8.1. Effect of increasing the mesh size in the North Sea trawl fishery for roundfish from 85 to 120 mm. A. Long-term change in SSB. B. Long-term change in landings (courtesy of Henrik Gislason).

Gislason (1993) projected the outcome of a 25% reduction in fishing effort in the roundfish fleet. He found that these projections depended on assumptions about the level of recruitment of each species. Particularly for haddock, sprat, and sandeel, the direction of change depended on assumed recruitment levels.

These early studies did not incorporate stock–recruitment relationships and may have underestimated the capacity of stocks to recover from overfishing. More recent projections have incorporated stock–recruitment relationships for each species. Collie *et al.* (2003) identified levels of fishing effort for which all ten North Sea species could be maintained at or above their respective precautionary levels. However, this scenario required substantial effort reductions in the trawl, industrial, and pelagic fleets. In these projections, prey species were able to escape predation control.

Since then, the multispecies work has continued using a range of other models focusing on species interactions such as Ecopath with Ecosim (Daskalov, 2002), Gadget (Howell and Bogstad, 2010) and SMS (Lewy and Vinther, 2004), models focusing on interactions in size based communities (Andersen and Pedersen, 2010) and models of economic trade-offs (Voss *et al.*, 2014). Being top–down controlled in their construction, these models have unanimously predicted that a decrease in forage fish abundance would follow from an increase in predator abundance if the productivity of lower trophic levels and harvest rates on forage fish remained unchanged.

8.2 Empirical evidence of what happened to stocks of prey species with rebuilding of predator species

Trophic cascades, the top–down control of community structure, have been researched and controversially discussed for decades (Frank *et al.*, 2005). While the number of studies examining top–down control in the oceans rapidly increased, empirical evidence has been sparse (Baum and Worm, 2009). It requires intense and long-term data collection of multiple trophic levels over large spatial scales, which limited such investigations for a long time. In addition there difficulties might occur in the discovery of top–down control effects since these are not uniformly strong in marine ecosystems; high predator functional diversity may dampen the strength of cascading effects while in other cases, prey responses might be inhibited by exploitation (Baum and Worm, 2009).

Most of the available empirical evidence results from investigations of prey responses to predator declines. In contrast, there are few examples of management measures succeeding in reversing predator depletion. Some authors even suggest that predator recovery could be inhibited by the large increases in prey biomass, since the success of large predatory fishes may depend on adults cropping down forage fishes that are predators or competitors of their young (Swain and Sinclair, 2000). For example, in the Baltic Sea an increase in pelagic fish biomass was thought to hamper cod recruitment success due to clupeid predation on cod eggs (Köster and Möllmann, 2000).

8.2.1 Predator rebuilding

8.2.1.1 Barents Sea

Since 2003, the commercial fish stocks, in particular the gadoids, of the Barents Sea have recovered from below B$_{PA}$ to historically high levels. The cause was probably a combination of environmental conditions and a new management regime. Increased temperature led to higher zooplankton production, a wider area for foraging and higher growth rates (Ottersen and Loeng, 2000; Johannesen *et al.*, 2012). At the same time Norway imposed a ban on discarding and, together with Russia, a strong monitoring of the fishing fleet, which has led to reduced illegal fishing and unreported landings.

In the 1980s and 1990s, the fish stocks in the Barents Sea fluctuated widely. The capelin (*Mallotus villosus*) collapsed to 5% of the previous biomass in 1986, leading to reduced growth rate skipped spawning and cannibalism in its predator, the northeast Arctic cod (*Gadus morhua)*, (Hamre, 1994; Gjøsæter *et al.*, 2009; Skjæråsen *et al.*, 2012). Further, seabirds, especially thick-billed murre (*Uria lomvia)*, died of starvation, sea mammals such as minke whale (*Balaenoptera acutorostrata*) lost weight and harp seals (*Pagophilus groenlandicus*) migrated to the coast of mainland Norway (Vader *et al.*, 1990; Barrett og Krasnov, 1996; Haug *et al.*, 1995; 2002). Although fisheries were claimed to be the cause, research revealed that the loss of capelin was mostly due to a combination of predation by and possibly prey competition with strong cohorts of young herring benefitting from increased temperature, depleting the recruitment of the capelin (Gjøsæter and Bogstad, 1998; Hallfredson and Pedersen, 2006; 2007). Recruitment failure in a semelparous species like the capelin will have dramatic effects on the stock. It seems that the Barents Sea ecosystem is shaped in large by forage fish abundance, fitting the "wasp-waist" paradigm whereby top predators are controlled by their prey rather than the reverse situation (Hjerman *et al.*, 2010).

More capelin collapses followed, in 1993 and 2003, lasting for up to five years, probably prolonged by continuous fisheries on the remaining stock. However, the strong cohorts of herring in the same periods provided the cod stock and thereby other top predators with food. The increased stock of young cod sustained sea mammals and starvation was avoided. In the years since 2006, the cod, NEA haddock (*Melanogrammus aeglefinus*) and Northeast Arctic saithe (*Pollachius virens*) stocks have developed to record high levels. Still, the TAC for each stock quota is set based on considerations of the demands and pressure on all the commercial species in the ecosystem, including sea mammals. Models are applied to assess the possible trends of the capelin, young herring and cod stocks, from theoretical and empirical data concerning prey availability and the predation pressures on the different stocks (Lindstrøm *et al.*, 2009; Ottersen *et al.*, 2014). The aim for the fishery management is to keep fishing pressure at a level where the resources removed do not impair the natural functioning of the ecosystem.

The recovery of the gadoid stocks has been unprecedented. Since the capelin stock recovered after the 2003 collapse, the stock has been fluctuating, but always above the precautionary threshold although gadoids have reached record high abundance levels. Recovery of sea mammals is not reported and seabirds, especially the thick-billed murre, show little signs of recovering. The ringed seal (*Phoca hispida*) is expected to respond with decreased reproduction in lack of ice (Kovacs *et al.*, 2014). Even the harp seal (*Pagophilus groenlandicus*) in the Spitsbergen area are now facing limitation in breeding grounds and increased surveillance is suggested to look for negative consequences also for this species (Anne Kirstine Frie, IMR, pers comm. The future development of the Barents Sea is followed closely by annual surveys and modelling. However, at present there is no evidence of a negative effect of the large predatory stock on prey fish biomass, probably due to the lasting strength of the capelin stock.

8.2.1.2 Georges Bank

The Georges Bank haddock (*Melanogrammus aeglefinus*) stock rebuilt rapidly from low levels in the 1990s to high levels in the 2000s that had not been observed since the 1960s (Figure 8.2). It is known from diet data that haddock prey on the demersal eggs of herring (*Clupea harengus*). Data from ichthyoplankton surveys indicate that herring egg survival declined as predation pressure from haddock increased (Richarson *et al.*, 2013). A population model fitted to these data suggests that herring can exist at a high and low equilibrium level, depending on the levels of predation and fishing mortality. With higher levels of haddock predation, a lower fishing morality on herring may be required to maintain the stock at the upper equilibrium. Hence, the two species showed signs of both a decreased survival of prey fish as the predator abundance increased and of the need to decrease fishing mortality as natural mortality increased.
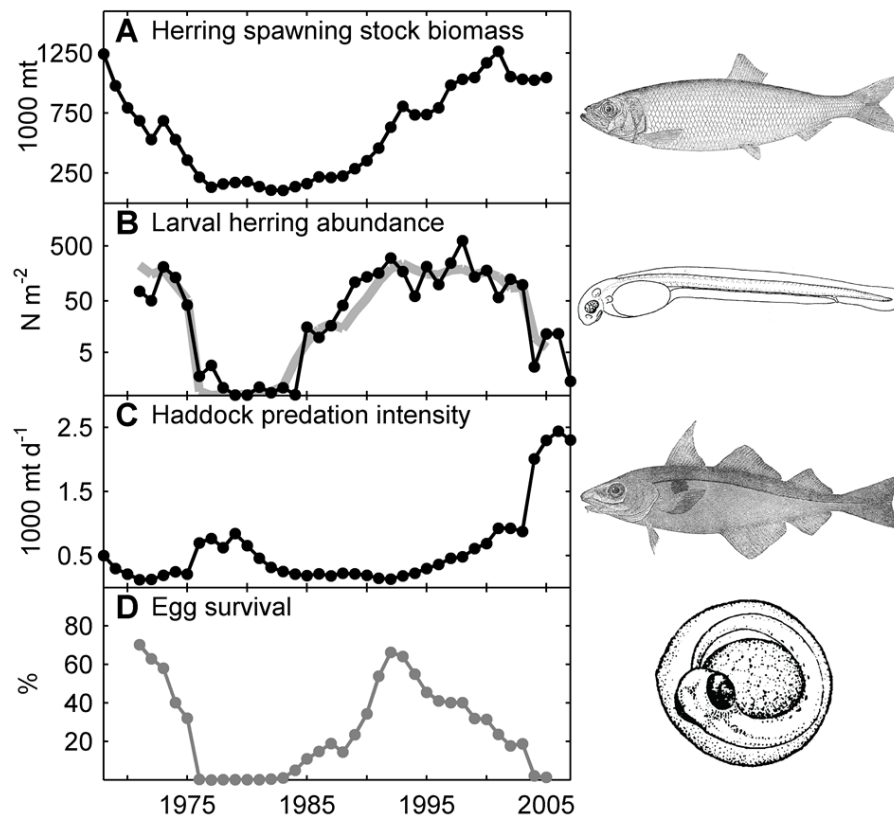
Figure 8.2. Effect of predation by Georges Bank haddock on the egg survival of Atlantic herring. From Richardson *et al*. (2011).

8.2.1.3 Baltic Sea

The eastern Baltic cod has gone through a time-series of serious decline followed by a recent rebuilding of the stock and hence is another example of successful recovery of a predator species. However, there were several unforeseen aspects of the recovery. Firstly, the distribution of the rebuilt stock remained the same as when depleted and hence only part of the historical habitat appears to be occupied (Eero *et al*., 2012). In the occupied habitat, declines in forage fish compared to other areas have been observed, presenting an example of likely top–down control. Following this decline in prey fish, evidence of bottom–up control of predator growth rate has emerged as the cod weight-at-age has decreased to historically low levels (Figure 8.3).
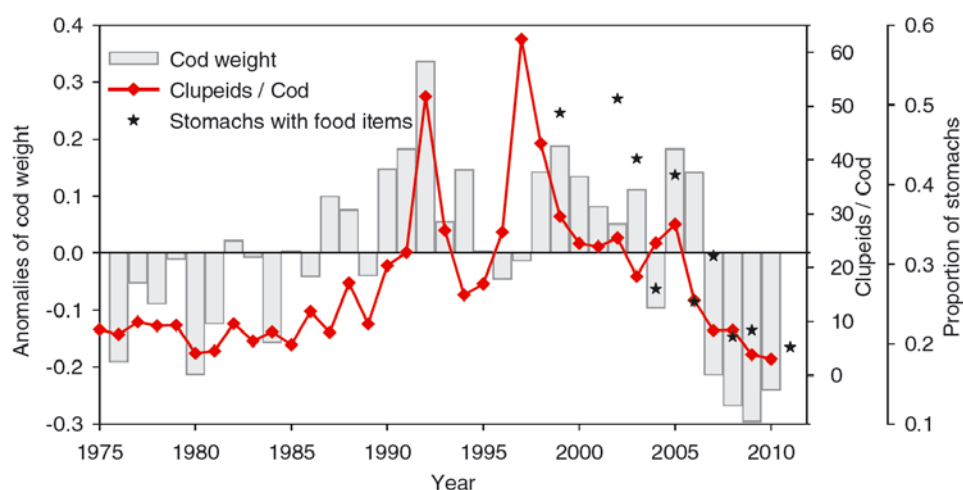
Figure 8.3. Anomalies in mean weight of cod (average of age groups 4–7) in Subdivision 25 of the eastern Baltic Sea (bars) compared with changes in the biomass of clupeids (sprat and herring) relative to the number of adult cod (at age 4 and older) in the same area (line). The stars show the proportion of cod stomachs containing food items. From Eero *et al*. (2012).

8.2.2 Response of prey species when predator species were depleted

There are many examples of prey increases due to declining abundances of predatory fish stocks: e.g. the semi-enclosed Baltic Sea system (Harvey *et al.*. 2003; Möllmann *et al.*, 2008; Österblom *et al.*, 2007), the enclosed ecosystem of the Black Sea (Daskalov, 2002), the open ocean ecosystem of the Central Pacific (Ward and Myers, 2005) as well as in continental shelf ecosystems of the North Atlantic (Worm and Myers, 2003), the Northwest Atlantic (Frank *et al.*, 2005; Fogarty and Murawski, 1998) and Northeast Atlantic (Blanchard *et al.*, 2005; Daan *et al.*, 2005; Sparholt *et al.*, 2002). In most cases prey populations were found to increase with predator depletion, but compensating effects among predators were also discovered (Fogarty and Murawski, 1998). Worm and Myers (2003) used meta-analysis to explore predator–prey interactions between Atlantic cod (*Gadus morhua*) and its prey the northern shrimp (*Pandalus borealis*) in several regions across the North Atlantic Ocean. Their findings show evidence of strong top–down effects recognizable in fishery induced decreases in predator biomass leading to increased prey populations. Rebuilding will likely follow a different trajectory than predator depletion, but these examples may indicate the strength of top–down predator control.

8.2.2.1 North Sea Norway pout

Cod, whiting and saithe are by far the main predators on Norway pout (*Trisopterus esmarkii*) in the North Sea. During the decrease in stock sizes of the three main predators from 1974 to 1999, mortality estimates obtained directly from survey analysis showed a clear decrease consistent with the decline in predator populations (Sparholt *et al.*, 2002; Figure 8.4). This is consistent with a top–down control of mortality responding to a release in predation. The analysis was possible because the surveys of Norway pout have very high internal consistency (strong cohorts turn up in consecutive surveys) and has to our knowledge not been repeated for other species.
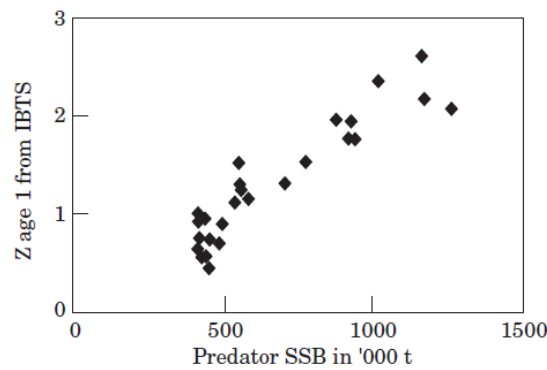
Figure 8.4. Total mortality (Z) of age 1 Norway pout from IBTS data plotted vs. spawning–stock biomass (SSB) of cod, whiting and saithe combined (1974–1997). From Sparholt *et al.* (2002).

8.2.2.2 Eastern Baltic cod

In the Baltic Sea ecosystem Österblom *et al.* (2007) investigated the dynamics of the main fish stocks, cod (*Gadus morhua*), herring (*Clupea harengus*) and sprat (*Sprattus sprattus*) between 1900 and 1980. They identified seal hunting as an important driving force reducing seal abundances and thereby allowing cod to become the dominant predator in the ecosystem at the beginning of the last century. Prior to the recent re-building of the eastern Baltic cod stock, the stock exhibited a long-term decline in abundance (Casini *et al.*, 2009). This decline was followed by a concurrent decrease in mortality of the main forage fish species which subsequently increased dramatically in abundance (Figure 8.5), consistent with a release from predation mortality and hence top–down control. Harvey *et al.* (2003) further identified changes in cod bio-mass affecting benthic macrofauna which in turn caused inverse responses in benthic meiofauna.
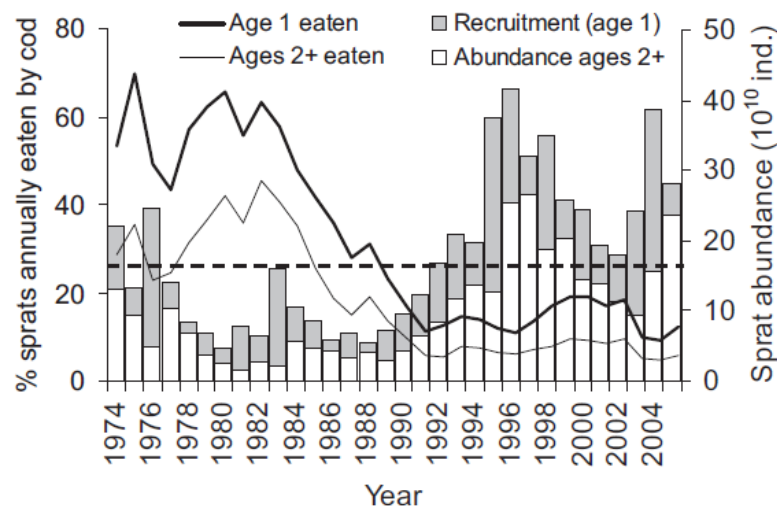

Figure 8.5. Trends in annual sprat predation mortality and sprat abundance. The columns represent sprat total abundance divided into recruits (age 1) and older individuals (ages 2+). The lines show the trends in the proportion of sprats that are eaten annually by cod (proportion of age *t* sprats that die from age *t* to age *t*+1 because of cod predation). From Casini *et al.* (2009).

8.2.2.3 Eastern Scotian Shelf

Frank *et al*. (2005) investigated the effects of fisheries removal of large benthic predators in the eastern Scotian Shelf ecosystem off Nova Scotia, Canada. They provided evidence of a trophic cascade involving the entire community over four trophic levels and nutrients, with the main transition period during the mid-1980s and early 1990s. This cascade was driven by declining abundance of large predators, mainly cod (*Gadus morhua*), but also haddock (*Melanogrammus aeglefinus*), white hake (*Urophycis tenuis*), silver hake (*Merluccius bilinearis*), pollock (*Pollachius virens*), cusk (*Brosme brosme*), redfish (*Sebastes* spp.), American plaice (*Hippoglossoides platessoides*), yellowtail flounder (*Limanda ferruginea*), thorny skate (*Raja radiata*), and winter skate (*Raja ocellata*). As a result of the collapse of the demersal fish community, the abundance of its primary prey, small pelagic fishes and benthic macroinvertebrates such as northern snow crab (*Chionoecetes opilio*) and northern shrimp (*Pandalus borealis*) increased markedly.
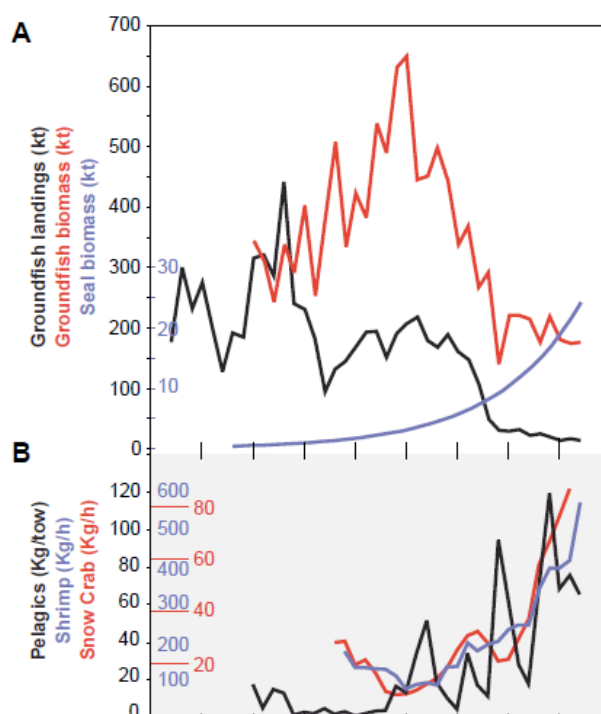

Figure 8.6. Illustration of a trophic cascade on the eastern Scotian Shelf across four levels and nutrients. (A) Commercial landings of benthic fish species, fishery-independent survey estimates of benthic fish, and population biomass estimates of grey seals. (B) The forage base of benthic fish species (and seals), including small pelagic fish species and benthic macroinvertebrates. From Frank *et al*. (2005).

8.2.2.4 Gulf of Maine

The northwestern Atlantic ecosystem outside the coast of New England, USA and southeastern Canada has changed dramatically since the mid-1900s (Steneck *et al*., 2004 and references therein). Large fish species had functioned as top-predators in a stable ecosystem relationship for at least 4000 years until the early 1900s, when mechanized technology and engines in the vessels led to increased fishing efficiency (Figure 8.7). By 1960, large fish were rare along the coast in the Gulf of Maine and invertebrates, especially the echinodermata, like the sea urchin *Strongylocentrotus droebachiensis* grazed down the kelp in protected areas and encrusting calciferous algae covered hard bottoms in more exposed areas, leaving little sheltering habitats to

the stages of juvenile fish. Lack of top predators led to a steady-state ecosystem, dominated by sea urchins for more than 20 years, when intensive sea urchin harvesting was initiated in the late 1980s. The stock of sea urchins was then reduced to below carrying capacity in less than ten years and the kelp cover recovered in less than three years from the removal of the sea urchins. Although protection measures were introduced in the late 1990s, the kelp cover has remained. By 2000 the sea urchins were controlled by increase in the stocks of decapod crustaceans, particularly the American lobster (*Homarus americanus*). This ecosystem with decapods as major top predators has seemed stable since the mid-1990s.
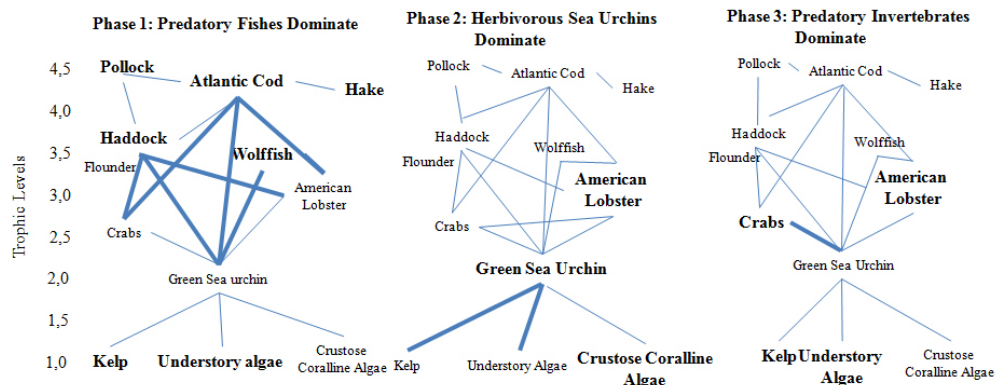

Figure. 8.7. Foodwebs in the Gulf of Maine from three phases. Phase 1: From historic times to the 1970s, with large fish top predators. Phase 2: From the 1970s to mid-1990s: Encrusting coralline algae and sea urchins. Phase 3: from mid-1990s to present: The ecosystem dominated by large decapods. Based on figure from Steneck *et al.* (2004).

8.2.3   Additional factors affecting prey species dynamics

Prey species face other problems besides predation pressure from commercial fish species. Several stocks of prey species have shown declines in recruitment and/or growth which cannot be explained by top–down control. For example, Baltic Sea and Celtic Sea herring have exhibited severe declines in growth and condition that could be due to food limitation (Figure 8.8, Casini *et al.*, 2010).
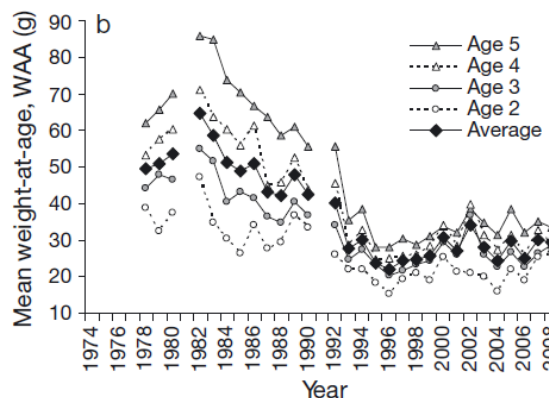

Figure. 8.8. Mean weight-at-age of central Baltic herring. From Casini *et al.* (2010).

In the Barents Sea surveys of 2012–2013, reduced recruitment in NEA haddock has been recorded since 2006 after three years of record high levels and the SSB is still

very high. The reasons for the assessed reduction in recruitment are not clear. It may or may not be a result of the increase in the cod stock, since uncertainty in the stock assessment due to inconsistent landing reports and unreported discards and, environmental drivers must also be considered. Low haddock recruitment may constrain the cod fishery because cod and haddock are caught by the same fleets.

In the North Sea, non-commercial top predators such as marine mammals and non-target fish species are increasingly exerting predation pressure particularly on gadoids (WGSAM, 2011). Similar evidence of increasing predation pressure has been seen on the Canadian east coast (Chouinard *et al.*, 2005). These other processes confound the top–down effects of piscivorous fish on their prey species.

8.3 Projections of the effects of rebuilding from multispecies models of different regions

There are operational multispecies models for many ICES areas and in several cases, these models have been used to predict the effect of predator rebuilding. The general structure of these models conforms to the theory of a top–down controlled system because predators exert an influence on prey which in return exert either no or a substantially smaller effect on predators. Despite these built-in assumptions, these models are still useful to evaluate the amplitude and direction of the resulting response, as this is not easily predicted in complex foodwebs.

8.3.1 Indirect effects and trophic cascades

Indirect effects and trophic cascades occur frequently in top–down controlled models (WGSAM, 2012) and size-spectrum models (Andersen and Pedersen, 2010). The number of levels depends on the number of modelled trophic levels below the predator and the amplitude of the effects vary from slight to moderate. Here we divide the indirect effects of rebuilding predators into the effect on the prey of the predator and effects of removing prey on other components of the ecosystem. Effects of the predator on the prey are seen in all ecosystems and the effect is always an increase in predation mortality with increasing predator abundance (WGSAM, 2012; WKBALT, 2013; Curti *et al.*, 2013). However, effects of this reduction on the remaining parts of the ecosystem are more variable. In the North Sea, reducing fishing mortality on cod is expected to decrease the abundance of whiting and haddock, which in turn would increase the abundance of their prey, the forage fish (WGSAM, 2012). So if all other *F*s were kept constant, the net effect of increasing cod would be to increase forage fish abundance. This increase in forage fish in turn decreases the effect of predators on forage fish on their alternative prey. In general, prediction of this type of effects is difficult and subject to a larger uncertainty about the direction and magnitude than the effect on the prey of the predator that increases in abundance.

8.3.2 Magnitude of change

Though the direction of the anticipated change in prey biomass with increased predator abundance is relevant, it is really the magnitude of change that is of interest to managers. Given all the other factors affecting prey stocks, a decrease of forage fish of say less than 5% is unlikely to have a noticeable effect on the other components of the ecosystem.

The magnitude of increase in forage fish observed concurrent with a severe decrease in predatory fish ranges from slightly more than a doubling of biomass on the eastern Scotian Shelf (Frank *et al.*, 2005) to an approximately threefold increase in sprat bio-

mass (Casini *et al.*, 2009) and a decrease of natural mortality of Norway pout of about 75% (Sparholt *et al.*, 2002). In contrast, there was virtually no change in local sprat abundance in the eastern Baltic Sea when cod biomass was rebuilding (Eero *et al.*, 2012).

In comparison, the magnitude of the predicted change in herring and sprat SSB in the Baltic Sea as cod fishing mortality is increased from 0.4 (close to F_{MSY}) to 0.7 (around or above F_{PA}) is only up to 20% (WKBALT, 2013). In the North Sea, the foodweb is more complex, and trophic cascades are observed as described under 8.3.1. Here, a decrease in fishing mortality of cod from 0.6 to 0.5 leads to a 20% increase in SSB of haddock and whiting, both of which are preyed on by cod. However, the forage fish, which are preyed on by the now scarcer haddock, whiting and juvenile cod, are released from predation and exhibit increases in SSB between 5 and 20%. These large differences were seen with a very modest decrease in fishing mortality well below the decrease required to reach the currently used F_{MSY} of North Sea cod of 0.19 (WGNSSK, 2013). Hence, the effects on the prey stocks appear to be greater rather than smaller in the more complex North Sea model.

8.4 Evidence of food limitation in predators

Examples exist of food limitation in seabirds, marine mammals, and fish, mostly from temperate and boreal ecosystems. Predator condition may depend not only on food availability but also on food quality and availability to the predators. Conversely, there are many examples in which predator diets respond to changing prey abundance with no evidence of altered feeding rates, stomach content weight or growth rates (Engelhard *et al.*, 2014).

8.4.1 Evidence of food limitation in seabirds

The breeding success of kittiwakes at colonies on the southeast coast of Scotland provides one of the clearest examples of a food limitation effect on annual chick production. Variation in chick productivity has been linked to variation in sea temperature in February and March with a one-year lag (Frederiksen *et al.*, 2004). The authors conclude that sea temperature exerts this influence on breeding success through mediation of the sandeel food supply. They speculate that warm winters reduced the abundance and growth of the current sandeel cohort, but kittiwakes only start to feed on, and provision their chicks, with this weakened cohort late on in the season; too late to have much effect on breeding success. But in the following year, this weak cohort, now 1-group fish, is inadequate to allow adult kittiwakes to get fully into breeding condition, and it is the poorer condition of adult birds one year after the warm winter that affects breeding success. A second study in the same location provides an almost identical result. Scott *et al.* (2006) show that kittiwake breeding success is positively related to the start date of the spring plankton bloom and the date of onset of stratification. They conclude that since kittiwakes are surface feeders, late stratification and a late plankton bloom help to maintain sandeel availability during the most critical period early in the breeding season. In both studies, the food limitation link between the environmental driver and chick productivity is speculative, but in both instances, when a sandeel fishery operated in the region, breeding success was reduced below the level expected from the relationship with the environmental driver (Figure 8.9). This reduction provides evidence that the removal of sandeels by the fisheries reduced sandeel availability to kittiwakes, resulting in bottom–up limitation on breeding success.
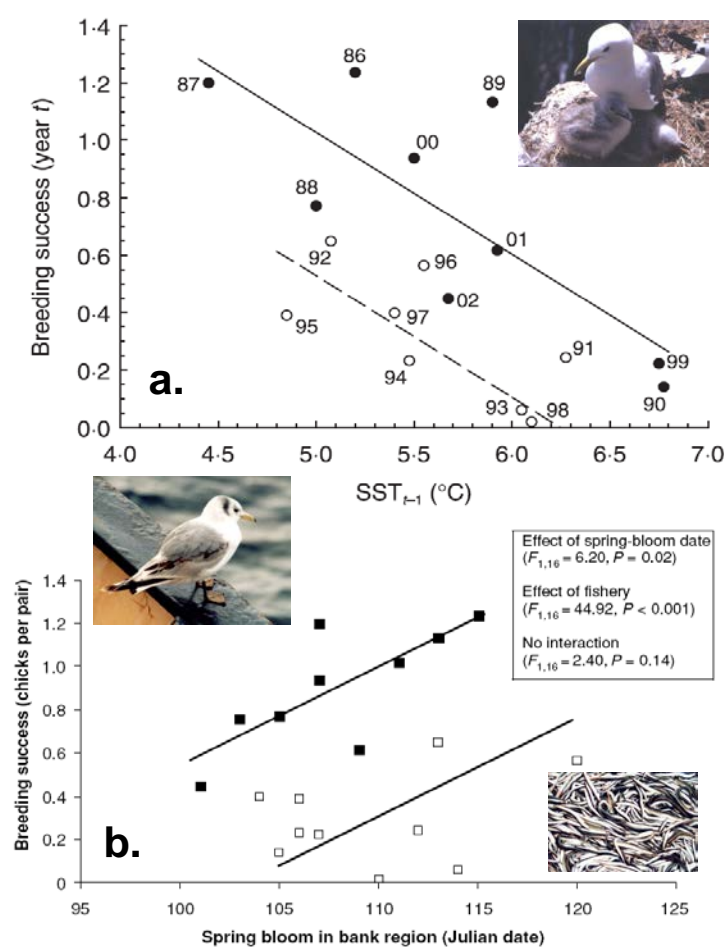
Figure 8.9. Relationships between (a.) spring sea temperature (with a one-year lag) (Frederiksen *et al.*, 2004) and (b.) the date of the onset of stratification and the start of the plankton bloom (Scott *et al.*, 2006) on kittiwake breeding success at colonies along the Scottish southeast coast. Relationships were estimated for years with (open symbols) and without (filled symbols) a sandeel fishery operating in the area.

From 1980 onwards, Norwegian spring-spawning herring *Clupea harengus* underwent a tenfold increase in stock biomass, which seems to have been associated with repeated collapses in the Barents Sea capelin *Mallotus villosus* stock. For several decades, while capelin abundance in the Barents Sea was high, the population of black-legged kittiwakes *Rissa tridactyla* breeding in northern Norway increased, but from the early 1980s onwards this trend has reversed, reaching a rate of -8% year^{-1} since 1995. In the southwestern Barents Sea, black-legged kittiwakes were dependent on capelin as prey in order for breeding to be successful. A positive relationship between diet composition and fish abundance, and a negative effect associated with switching to feeding on herring in the absence of capelin, on chick provisioning fledging success was observed (Barrett, 2007).

8.4.2 Evidence of food limitation in marine mammals

In the late 1980s, the Firth of Forth supported a major winter sprat fishery, but with declining sprat abundance, the fishery was closed in 1993 to protect juvenile herring from being taken as bycatch. The Moray Firth holds one of the largest concentrations of harbour seals *Phoca vitulina* around the UK coast. When sprat was abundant, harbour seals fed almost exclusively on sprat, maintaining high body condition through

to the following spring. At the start of the breeding season they were in good breeding condition and pup production was high. When winter sprat abundance declined, harbour seals were forced to feed on alternative prey. As the proportion of gadoid fish in the diet increased, adult body condition at the end of winter declined and pup production levels fell (Thompson *et al.*, 1996).

8.4.3   Evidence of food limitation in fish

Major cod fisheries exist in Arctic waters across the North Atlantic where capelin *Mallotus villosus* are their primary prey. When capelin stocks decline, as a consequence of either overfishing or lower recruitment, cod switch to alternative prey. In the Barents Sea, cod switched to the shrimp *Pandalus borealis*. Within a year or so, as a result of the higher predation mortality, the population biomass of shrimps began to decline. Unable to find sufficient alternative prey, rates of cannibalism among cod increased and cod food consumption rates fell. This, combined with a reduction in the quality of the prey consumed (alternative prey were of a lower energy density than capelin), resulted in reduced cod growth rates and lowered fecundity (Mehl and Sunnanå, 1991; Bogstad and Mehl, 1997; Bogstad and Gjøsæter, 2001). Modelling of the predator–prey interactions of cod and capelin in Icelandic waters suggested that when the capelin biomass dropped below 2 million tonnes, the consumption of capelin by cod fell to the point where cod were unable to compensate by consuming alternative prey. Total food consumption rates declined causing a reduction in cod growth rates, stock biomass and fisheries yield (Magnússon and Pálsson, 1991). Off southern Labrador and northeastern Newfoundland, cod were unable to consume sufficient alternative prey to compensate for the decline in their consumption of capelin in years when capelin biomass was low, suggesting that a sustained reduction in capelin stock would affect cod productivity (Lilly, 1991). In the North Sea, examples of food limited growth are scarce (Engelhard *et al.*, 2014).

8.5   Separating the effects of the landing obligation from effects caused by a move to MSY

This ToR was motivated by the concern that increased predation pressure on forage species could reduce their abundance and availability to dependent predators, including marine mammals and seabirds. At the same time, the introduction of a landing obligation is expected to lead to different fishery selection patterns. Reduced discarding could result in less food for some bird species. Rebuilding predator species could reduce the abundance of forage species for birds. Both measures could reduce the amount of food for some seabirds. But some seabird species depend more on discards and others on live prey, which means that the two processes could be distinguished in species-specific seabird responses.

8.6   Summary: what are the expected consequences for dependent predators?

Concerns about the potential indirect effects of rebuilding stocks of piscivorous fish have existed for several decades. These concerns have become more relevant as fishing mortality rate have been reduced toward their MSY levels. Still, there is unequivocal empirical evidence about the effect on prey species.

There is some support for the hypothesis of top–down control by predatory fish on prey fish. However, the evidence is equivocal, telling us that we should not always

expect these effects to happen. There is likely to be a substantial bias in the published literature, as lack of correlations are rarely reported in published manuscripts.

Review of the existing examples indicates that prey species generally decline when their predators increase but this response may be mediated by other processes. As predator stocks rebuild, they may become increasingly food limited and the rebuilding rate may slow down. Prey species may vary due to other factors, such as food limitation, independently of predation levels. Even if prey species decline with increased predation, they can still be maintained above precautionary levels, especially if fishing mortality on prey species is conditioned (reduced) on predator abundance. Multiple equilibrium levels of prey populations are possible, in which case reduced fishing pressure may be needed to maintain prey species in the domain of attraction of their upper equilibrium levels with increased predation pressure.

Evidence exists that some seabird and mammal species are food limited. The indirect effects of rebuilding piscivorous fish species on dependent seabird and marine mammal populations are difficult to predict. As the number of links between piscivorous fish and dependent predators increases, even the sign of the response may be unknown. The effects of rebuilding piscivorous fish may be amplified or damped, depending on the shape of functional responses (Rossberg, 2013), which in many cases are uncertain or unknown. Concerns about the indirect effects of rebuilding stocks of piscivorous fish do not provide compelling arguments for delaying rebuilding plans. Rebuilding stocks of large predator species will likely take ten years or more, whereas forage species respond on shorter time-scales. Thus fishing rates on forage species can be adjusted as their predator species increase in abundance.

8.7 Priority areas for study

- Conduct a structured meta-analysis of the incidence of top–down predator control in fish communities (and for seabirds) which includes null results to avoid reporting bias. This analysis could provide more general guidance on the expect effect of rebuilding predator species on their prey populations.

- Conversely, is bottom–up food limitation the exception or the norm? Investigate variation in feeding levels over time, for evidence of bottom–up control. This analysis would indicate whether fish stocks are likely to experience food limitation as they rebuild to MSY levels. WGECO proposes to continue the work on these two first topics under a dedicated ToR for 2015.

- Use population models to frame testable hypotheses about the strength of foodweb interactions, which can then be tested through field studies.

- In foodweb models, identify aspects that are data-limited (e.g. diet data, food selection models, and functional responses) to improve future knowledge.

- Better understanding of spatial distributions; overlap between predator and prey populations (e.g. Baltic Sea cod are in an area in which they do not overlap with sprat).

- Loss of stock structure, contraction of species to core habitats, changes in migration patterns, which may alter predator–prey interactions.

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9   Requests from other groups in the ICES system

9.1   Consequences of the plaice outburst on the North Sea ecosystem (WGNSSK)

This section is a response to the request from WGNSSK to WGECO: "According to WGNSSK estimates, the North Sea is currently ongoing a plaice outburst without precedent. However, plaice is not included in multispecies models, so the consequences of this outburst on the North Sea ecosystem are unclear and would potentially require additional focus".

9.1.1   Defining the "plaice outbreak"

Plaice spawning–stock biomass (SSB) has varied markedly over the last 56 years. Two earlier peaks, in 1967 and 1987, were both associated with exceptionally strong recruit cohorts produced two or three years earlier. Fishing mortality on plaice increased by a factor of over two between 1957 and 2001 and plaice catches (landings and discards) declined markedly from 1987 to 1996 because of a 59% decrease in SSB. From 2001, fishing mortality was reduced by 74%, to the lowest level ($F = 0.2$) in 2011. This has allowed a marked increase in plaice SSB, which for the last three years consecutively, has been higher than at any other time in the 56 y time-series (Figure 9.1). While there can be little doubt that this prolonged increase in plaice SSB has been sustained by the reduction fishing mortality, and the recent maintenance of fishing mortality at levels below those consistent with maximum sustainable yield ($F_{MSY} = 0.25$), it is also worth pointing out that the increase in SSB commenced two years after what was the seventh largest recruitment cohort observed within 56 y.

State–space "precautionary plots" relating annual steps in fishing mortality to annual changes in SSB show how increasing fishing pressure on the stock has progressively moved SSB away from the desired state, as inferred from current management reference points, and then how management has rectified this situation (Figure 9.2). These trajectories clearly show that, in recent years, the North Sea plaice stock has been in a situation unlike any other over the whole 56 year period. The MSY plot in particular shows that management targets for restricting fishing mortality have been met in the last five years (Figure 9.2a).
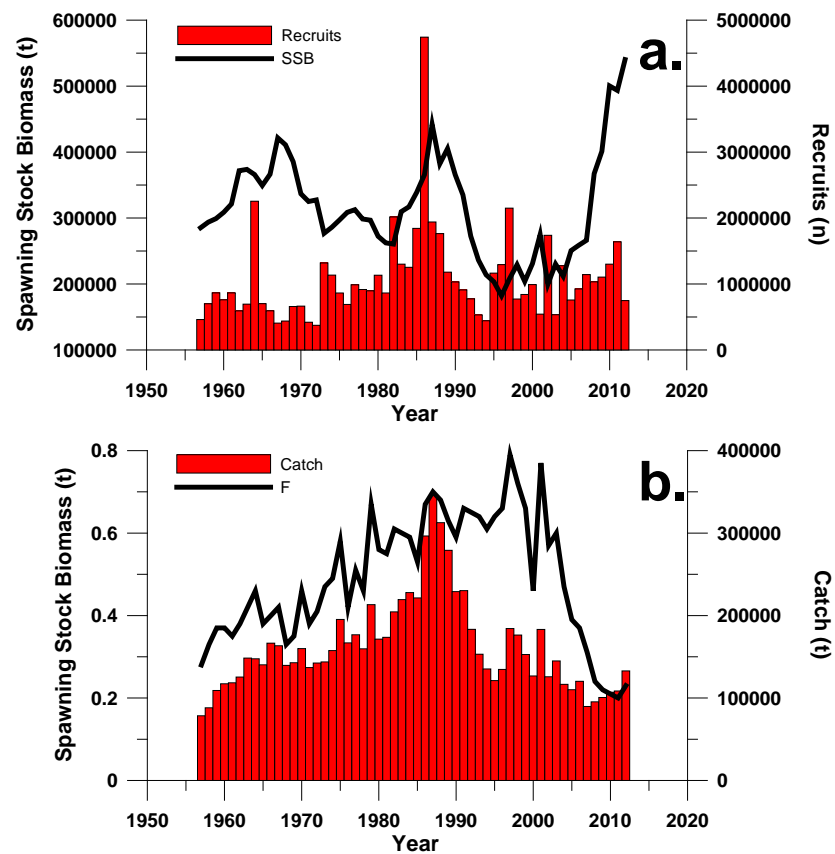
Figure 9.1. Trends in (a) plaice spawning–stock biomass (SSB) and recruitment strength and (b) plaice catches (human consumption and discards) and fishing mortality (F).
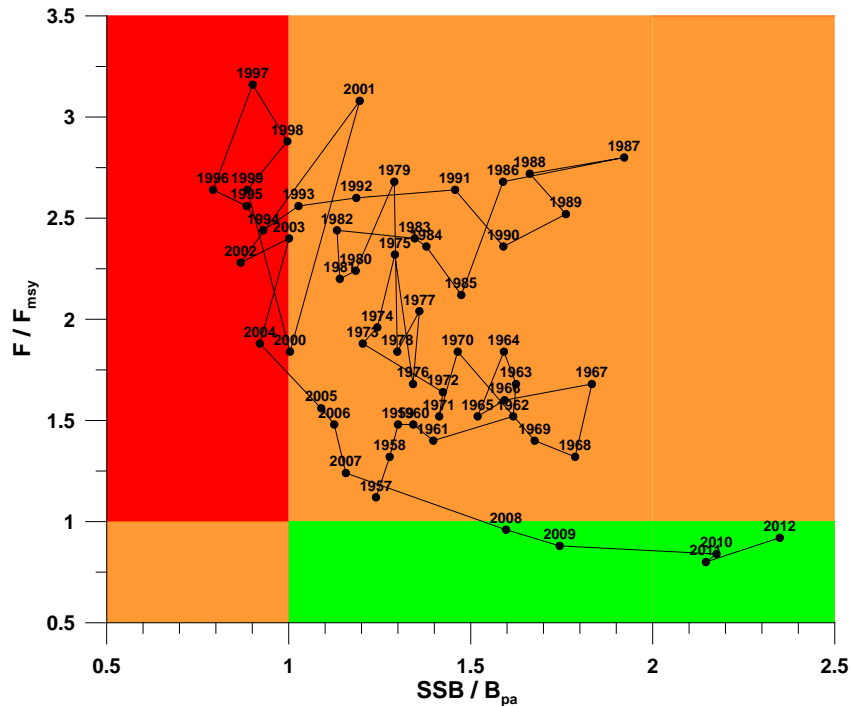
Figure 9.2. State–space "precautionary plots" showing the trajectory of the plaice population in two-dimensional space defined by criteria based on reproductive capacity, i.e. spawning–stock biomass (SSB) in relation to B_{MSY} (or actually its currently used proxy $B_{trigger}$ equal to B_{PA}) and fishing mortality (F in relation to F_{MSY}). Orange zones: one variable meets one criterion but not the other. Red zone: Status of the stock fails against both criteria. Green zone: Plaice status meets both criteria.

9.1.2 Trends in plaice and other demersal benthivore species biomass determined from survey data

Examination of the First Quarter (Q1) International Bottom-trawl Survey confirms the recent increase in plaice biomass, particularly since 2006. Again the data show that biomass estimates in the last three years of the time-series was higher than at any other time during the 29 year survey period. Several other demersal benthivorous fish show similar biomass trends (Figure 9.3). Common dab increased in biomass from 2001 and again biomass estimates in 2009 to 2011 were the highest three values observed in the 29 year data run. Lemon sole increased from 2008 and the 2011 data point was the highest in the time-series. Flounder have increased from 2002, and 2010 and 2011 were the highest estimates in the time-series. Grey gurnard have increased throughout the 29 year period and 2007 to 2011 constitute five of the six highest values observed. Lesser spotted dogfish have increased markedly since 1998 and biomass estimates determined for 2007 to 2011 are the five highest values in the time-series. Cuckoo ray biomass has increased since 1996 and 2010 was the highest value observed. Bullrout have increased since 2000 and the 2009 to 2011 estimates make up three of the six highest values observed (Figure 9.3).

Trends in the abundance of these eight species were examined in five subregions of the North Sea. The recent increase in plaice abundance was apparent in all five subregions (Figure 9.4). None of the subregional common dab trends matched the regional trend, but the regional trend was driven primarily by changes in common dab biomass in the Northeast, Southwest and Kattegat/Skagerrak subregions (Figure 9.5). The regional lemon sole biomass trend was mainly influenced by changes taking

place in the northeast and northwest North Sea (Figure 9.6). The regional flounder biomass trend was principally driven by changes taking place in the Kattegat/Skagerrak where biomass density was an order of magnitude higher (Figure 9.7). The increase in grey gurnard biomass density was apparent in all five subregions (Figure 9.8). Lesser spotted dogfish were scarce in the northeast North Sea and Kattegat/Skagerrak subregions, but the recent increase in biomass evident in the regional trend was apparent in all three of the remaining subregions (Figure 9.9). The regional trend in cuckoo ray biomass was only apparent in one subregion, the northwest North Sea, where biomass density was highest (Figure 9.10). Recent increases in bullrout biomass density were apparent in the four subregions where this species was most abundant, but in the remaining subregion, the northeast North Sea, no pattern was apparent (Figure 9.11). In conclusion, recent increases in the abundance of these species were generally evident across the entire North Sea and manifested in those subregions of the North Sea where each species was primarily distributed.
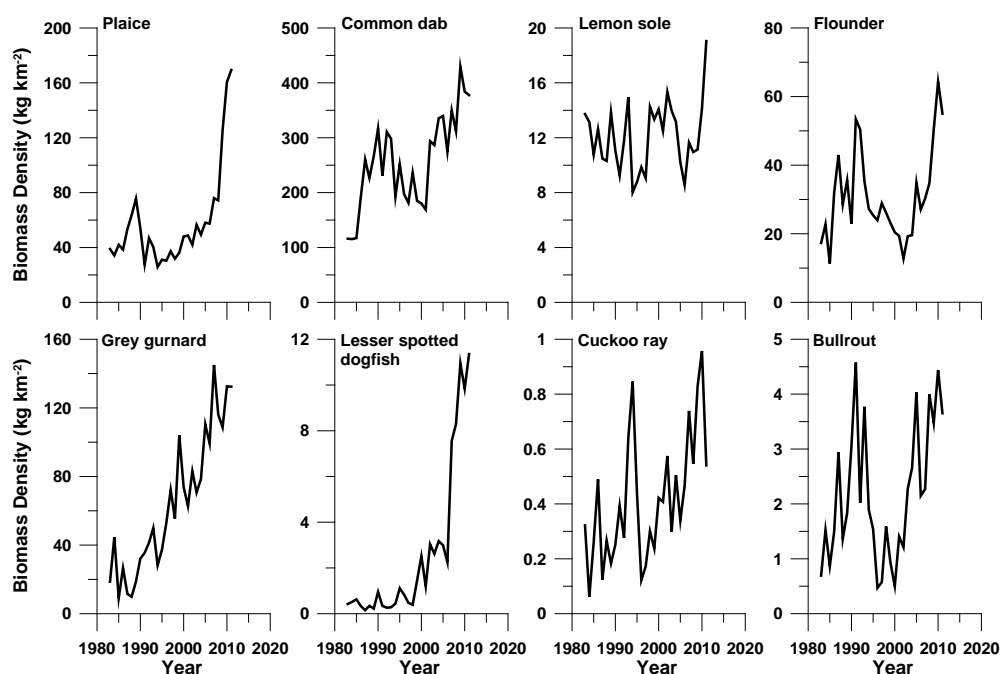
Figure 9.3. Variation in the mean biomass density of eight "demersal benthivorous" species determined from the First Quarter International Bottom-trawl Survey.

Figure 9.4. Variation in the mean biomass density of plaice in five subregions of the North Sea, the Northwest (NW), Northeast (NE), Kattegat and Skagerrak (KS), Southwest (SW), Southeast (SE), determined from the First Quarter International Bottom-trawl Survey.

Figure 9.5. Variation in the mean biomass density of common dab in five subregions of the North Sea, the Northwest (NW), Northeast (NE), Kattegat and Skagerrak (KS), Southwest (SW), Southeast (SE), determined from the First Quarter International Bottom-trawl Survey.

Figure 9.6. Variation in the mean biomass density of lemon sole in five subregions of the North Sea, the Northwest (NW), Northeast (NE), Kattegat and Skagerrak (KS), Southwest (SW), Southeast (SE), determined from the First Quarter International Bottom-trawl Survey.
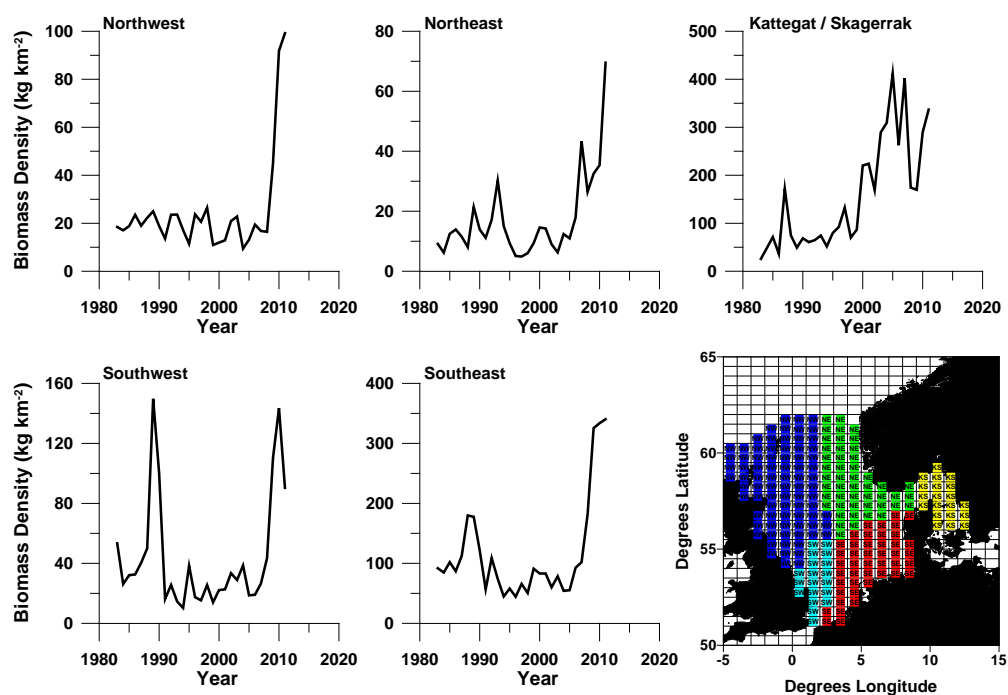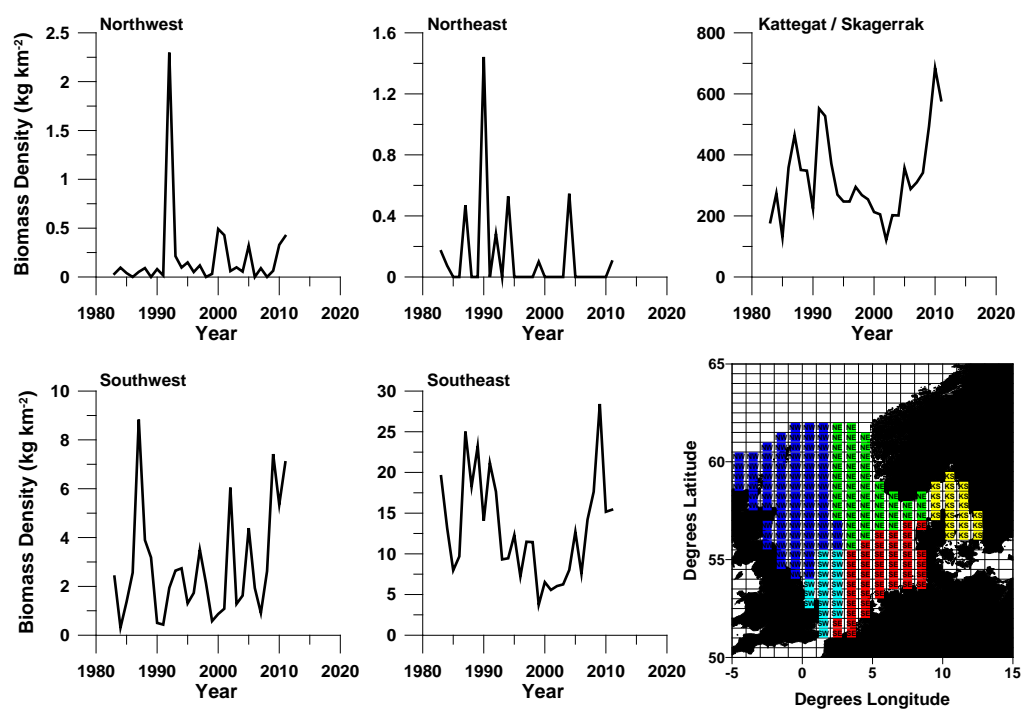

Figure 9.7. Variation in the mean biomass density of flounder in five subregions of the North Sea, the Northwest (NW), Northeast (NE), Kattegat and Skagerrak (KS), Southwest (SW), Southeast (SE), determined from the First Quarter International Bottom-trawl Survey.
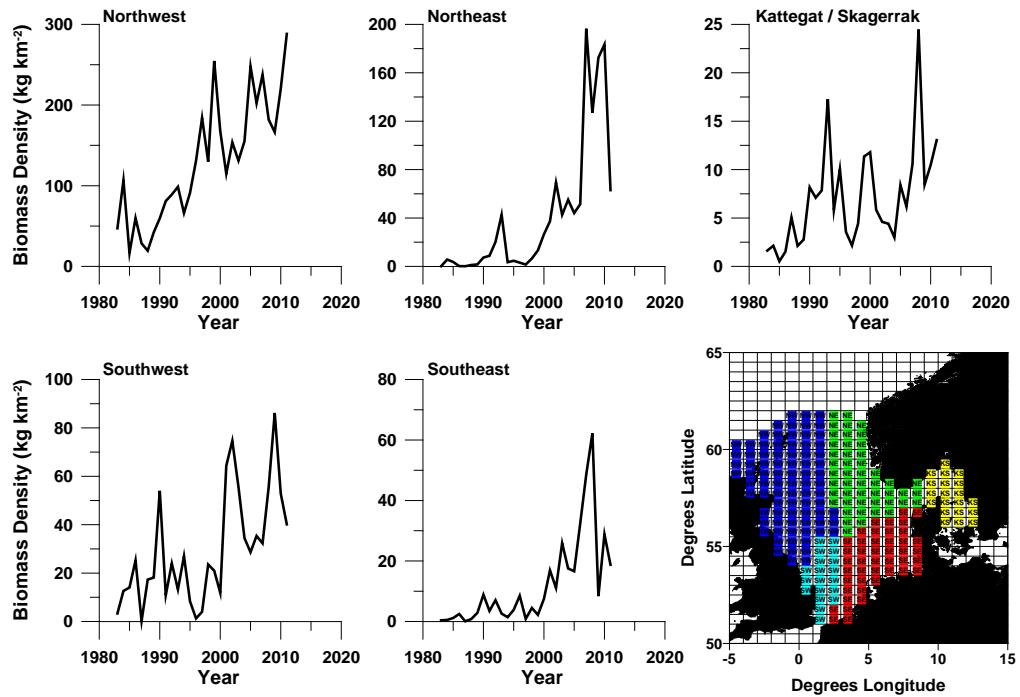
Figure 9.8. Variation in the mean biomass density of grey gurnard in five subregions of the North Sea, the Northwest (NW), Northeast (NE), Kattegat and Skagerrak (KS), Southwest (SW), Southeast (SE), determined from the First Quarter International Bottom-trawl Survey.
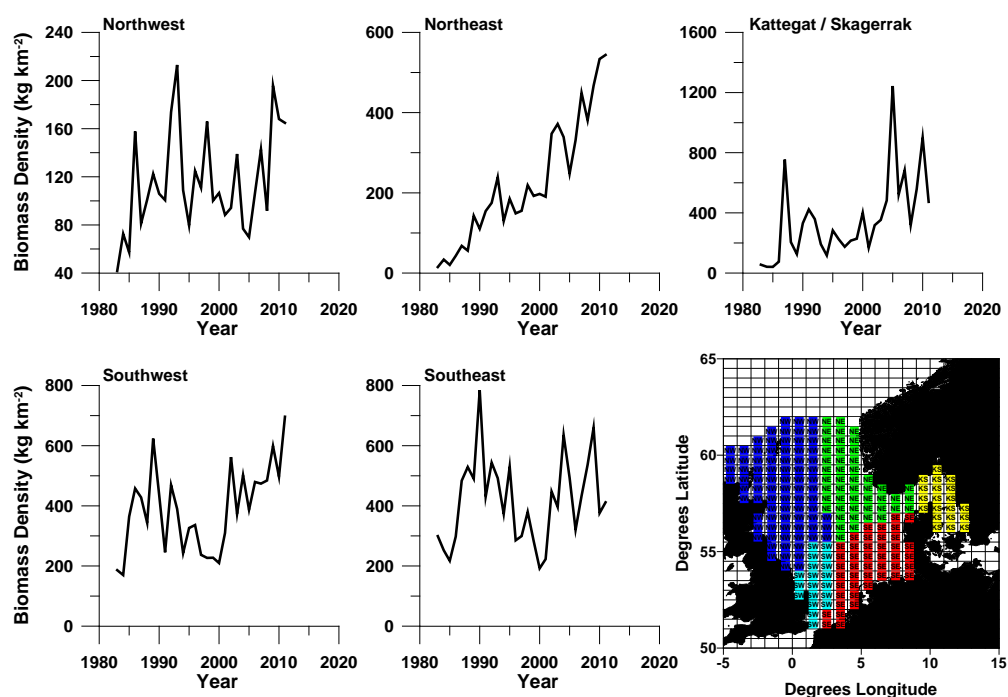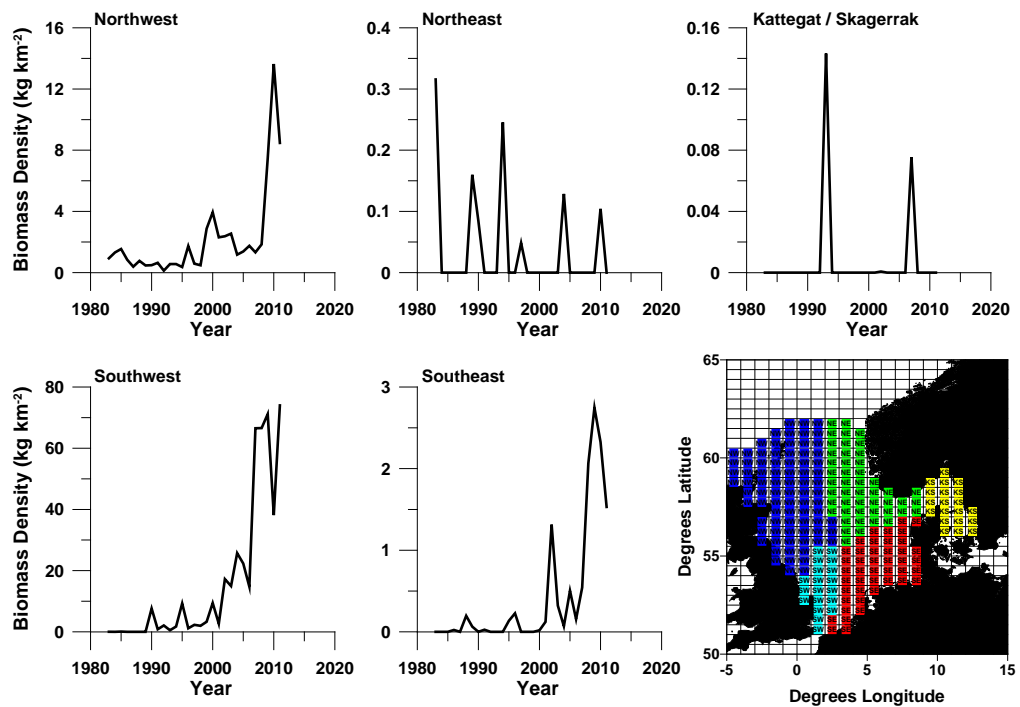

Figure 9.9. Variation in the mean biomass density of lesser spotted dogfish in five subregions of the North Sea, the Northwest (NW), Northeast (NE), Kattegat and Skagerrak (KS), Southwest (SW), Southeast (SE), determined from the First Quarter International Bottom-trawl Survey.

Figure 9.10. Variation in the mean biomass density of cuckoo ray in five subregions of the North Sea, the Northwest (NW), Northeast (NE), Kattegat and Skagerrak (KS), Southwest (SW), Southeast (SE), determined from the First Quarter International Bottom-trawl Survey.
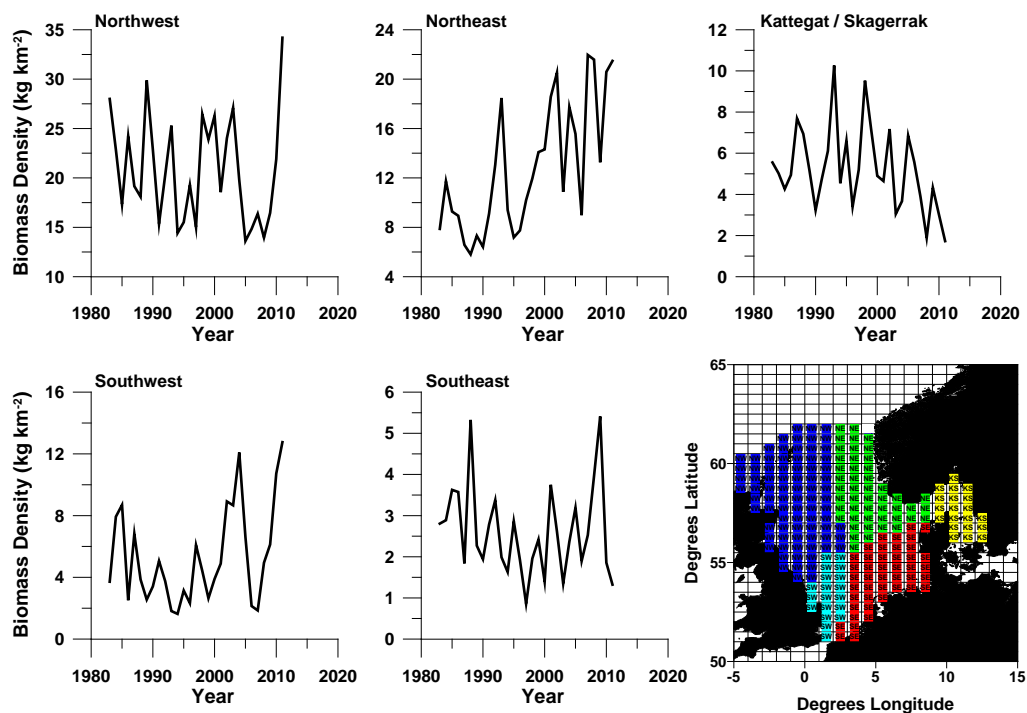

Figure 9.11. Variation in the mean biomass density of bullrout in five subregions of the North Sea, the Northwest (NW), Northeast (NE), Kattegat and Skagerrak (KS), Southwest (SW), Southeast (SE), determined from the First Quarter International Bottom-trawl Survey.
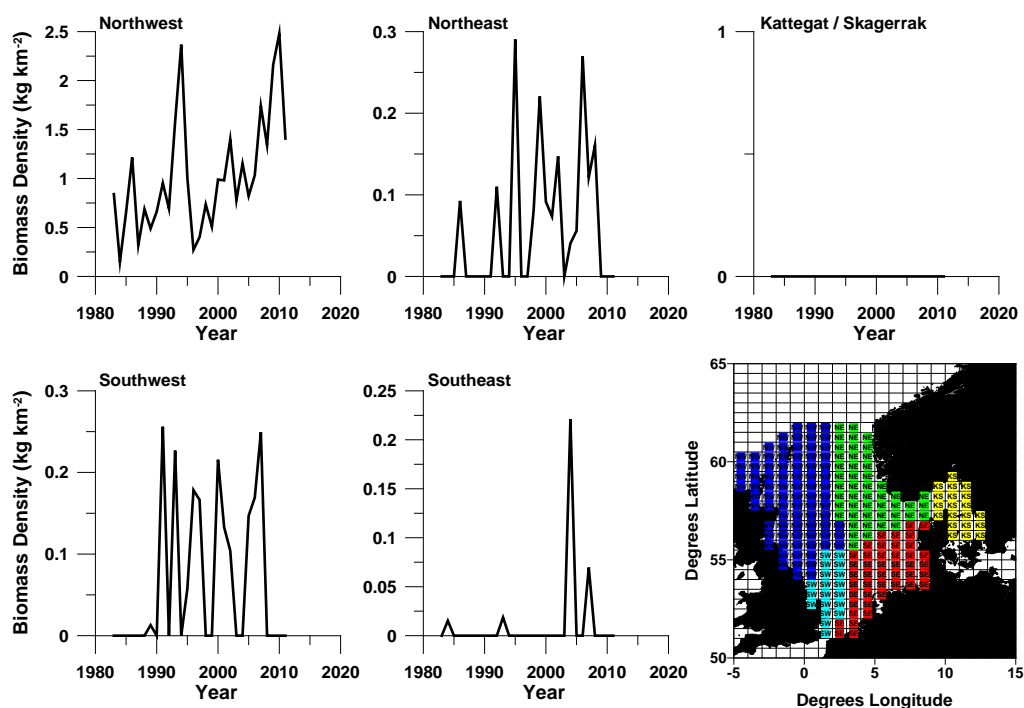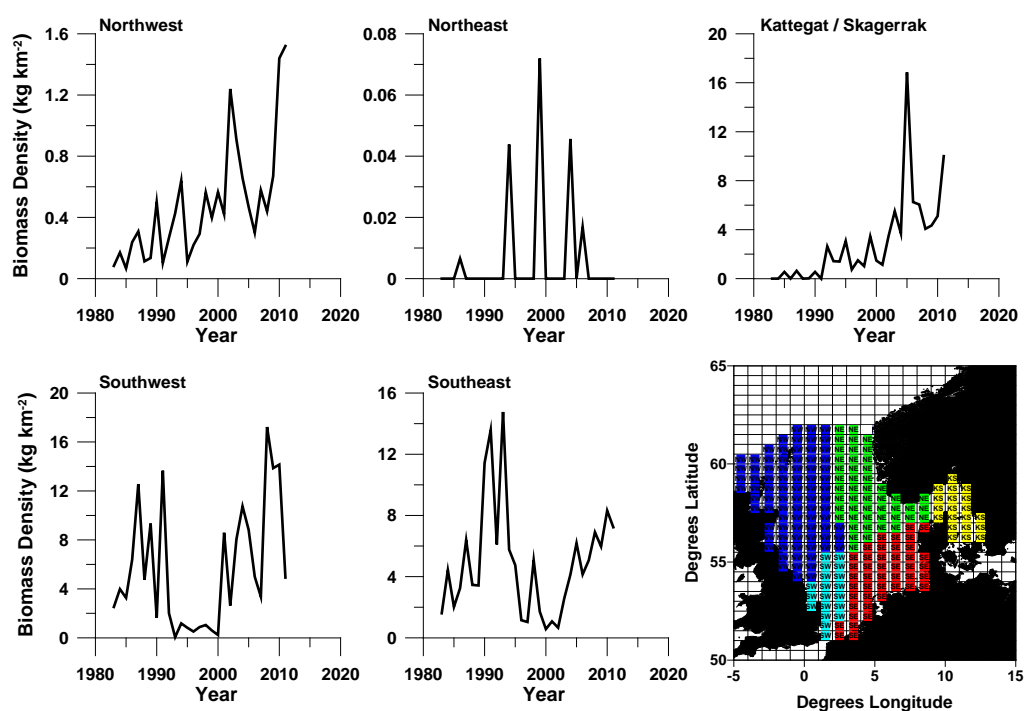
9.1.3 Trends in demersal benthivore biomass and predation pressure on benthic invertebrates

In Section 3, WGECO considers various foodweb indicators, including indicators of fish trophic functional guild biomass. Based on Q1 IBTS data, a twofold increase in demersal benthivore biomass was indicated over the 29 year period of the survey. This trend was derived using untreated survey data, but because all trawl gears sample the fish present in the trawled path imperfectly, not all fish present in the path are actually caught. Such guild biomass estimates therefore underestimate the actual biomass of demersal benthivorous fish present in the sea. If species composition of guild remains relatively constant, then changes in the guild biomass indicator can provide an indication of the relative change in actual guild biomass. But, here evidence is presented that suggests that major changes in the biomass of certain demersal benthivorous species have occurred; species composition of the demersal benthivore guild has certainly changed.

One solution to this problem is to apply catchability correction coefficients to the sample abundance-at-length data to derive estimates of absolute abundance in the sea. Fraser *et al.*, 2007 provide such coefficients for the each 1 cm length class of all species sampled in the third quarter (Q3) IBTS, and Figure 9.12 shows the trend in demersal benthivore biomass determined from these catchability corrected data. The transient effect of a strong juvenile haddock year class is obvious early on in this "catchability-corrected" Q3 IBTS time-series, but then the dominant effect of the increase in plaice biomass on estimates of absolute demersal benthivore biomass from 2004 onwards is clearly apparent. The overall influence of the increase in common dab biomass is down-weighted compared with plaice because the catchability of common dab in the GOV trawl is so much higher; correcting for catchability raises plaice biomass density estimates more than it raises common dab estimates.
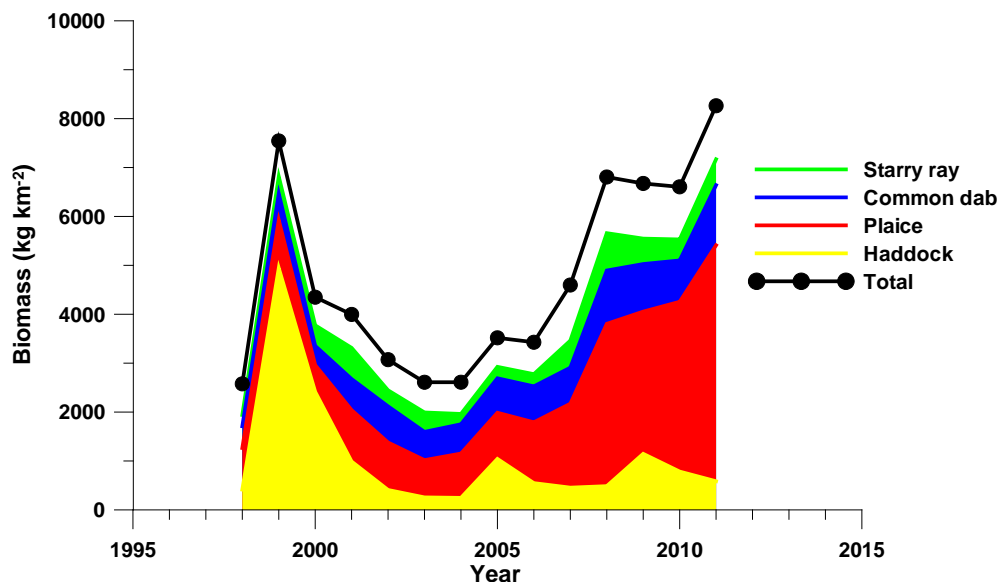
Figure 9.12. Variation in Demersal Benthivore trophic guild biomass derived from Q3 IBTS estimates of species biomass density at length estimates raised to take account of species- and size-related catchability in the GOV trawl.

Food consumption rates expressed as a proportion of fish body mass vary as a function of fish length, and flat fish intake rates tend generally to be higher than round-fish intake rates. Figure 9.13 shows the intake rates-at-length assumed for these two groups of fish along with the respective biomass-at-length–frequency distributions. These data were used to estimate predation loadings imposed by demersal benthivorous fish on the benthic invertebrate community (Figure 9.14a). A transient peak loading associated with the strong 1999 haddock year class quickly dissipated as these fish grew and became piscivorous, or were fished. From 2000 onwards the influence of the burgeoning plaice population on the benthic invertebrate community predation loading is clear. The percentage of this predation loading attributable to plaice and common dab increased appreciably (Figure 9.14b) and the relative change, using 2000 as the reference year, in plaice (Figure 9.14c) and combined plaice and common dab (Figure 9.14d) consumption was considerable. Considering only plaice, the predation loading on benthic invertebrates increased by a factor of 8.8 in just eleven years. Even when both species were taken into account, predation loading still increased by a factor of 6.

Figure 9.13. Food intake rates expressed as a percentage of fish biomass, assumed for each 1 cm length class of demersal benthivorous flat fish and round fish (a.) and biomass-at-length–frequency distributions of flat fish (b.) and round fish (c.) totalled across the full 14 y Q3 IBTS time-series.

Figure 9.14. Variation in daily consumption of benthic invertebrate prey by the whole demersal benthivore guild and by plaice and common dab (a.);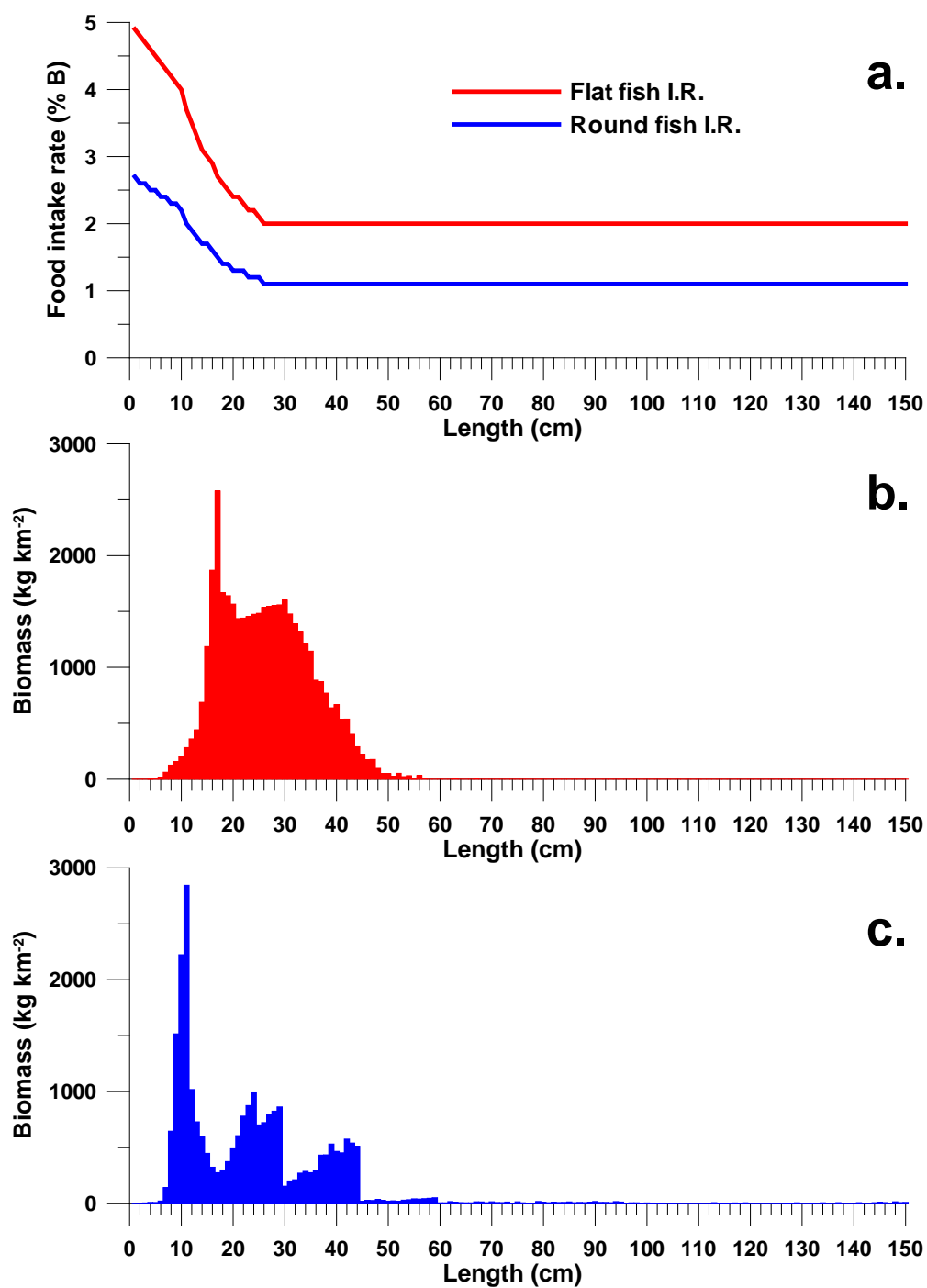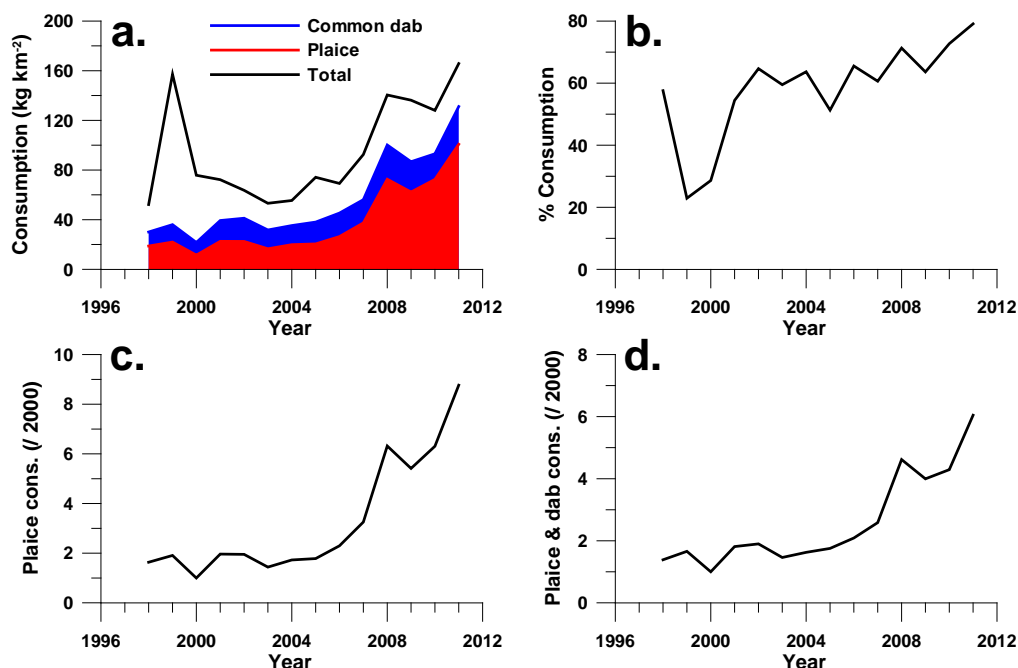 the percentage of all benthic invertebrate prey consumed each day consumed by plaice and common dab combined (b.); and the relative change in daily consumption of benthic invertebrate prey consumed by plaice alone (c.) and plaice and common dab combined (d.) using 2000 as the reference year.

9.1.4 Patterns in growth rate

The increase in the consumption of benthic invertebrate prey by the whole demersal benthivore guild, and particularly by plaice, raises the question as to whether the abundance of benthic invertebrate prey might be becoming limiting. If the biomass of demersal benthivorous fish is approaching its carrying capacity, then growth rates in the dominant species in the guild, in this case plaice, might start to decline. Figure 9.15 shows temporal variation in the weight-at-age in the stock of plaice, from age 1 to age 10. For ages 1 to 5 no obvious trend is apparent, but from age 6 to age 10 declining trends in weight-at-age are suggested, and these are particularly apparent in the very oldest aged fish. These data were sufficient to follow the full growth span, from age 1 to age 10, for the 1956 to 2002 cohorts (the 2002 cohort has age 10 in 2012). For each cohort, a growth coefficient was determined as the exponent of the power function of weight on age, both Log_{10} transformed. These growth coefficients showed a strong declining linear trend over the whole period (albeit with clear systematic variation in the residuals), and this has been related to increasing water temperature in the North Sea. However, fitting a 4th order polynomial function to the data suggested a marked decline in cohort growth towards the end of the time-series (Figure 9.16). This is perhaps indicative of plaice becoming food limited, possibly suggesting that B_{MSY} targets for the stock might be marginally too high to be supported by available benthic invertebrate food supplies. However, this evidence is by no means conclusive as polynomial functions are known to show a tendency for marked swings at the extremes of the data range. The situation will become clearer in a few years' time when data for more recent cohorts can be added to the analysis.
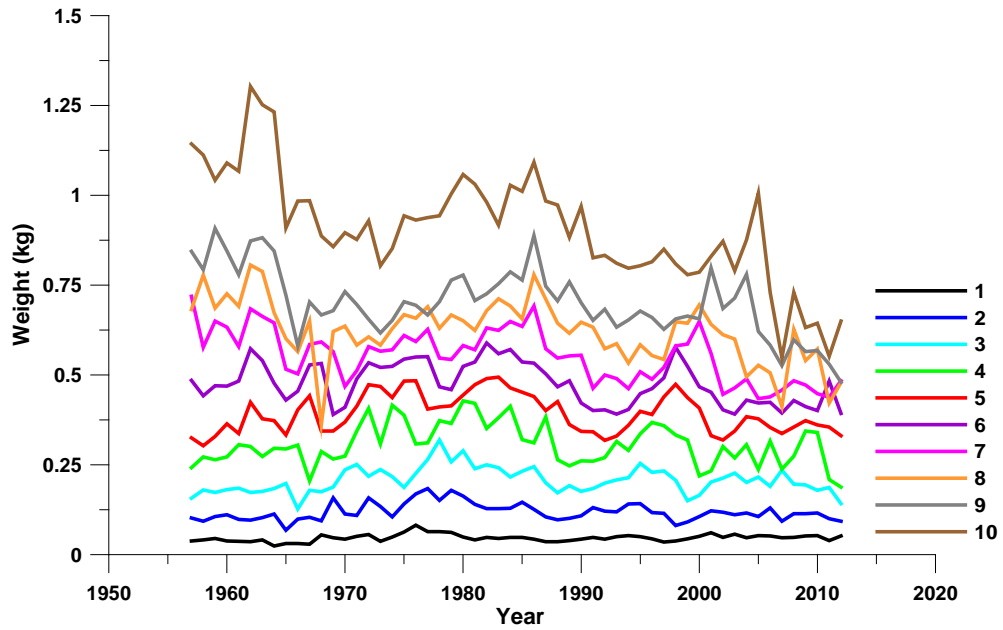
Figure 9.15. Temporal trends in North Sea plaice weight-at-age.

$$G = -2.667(10^7) + 5.398(10^4)C - 40.951C^2 + 0.01381C^3 - 1.746(10^{-6})C^4$$
$$r^2 = 0.531, n = 47$$
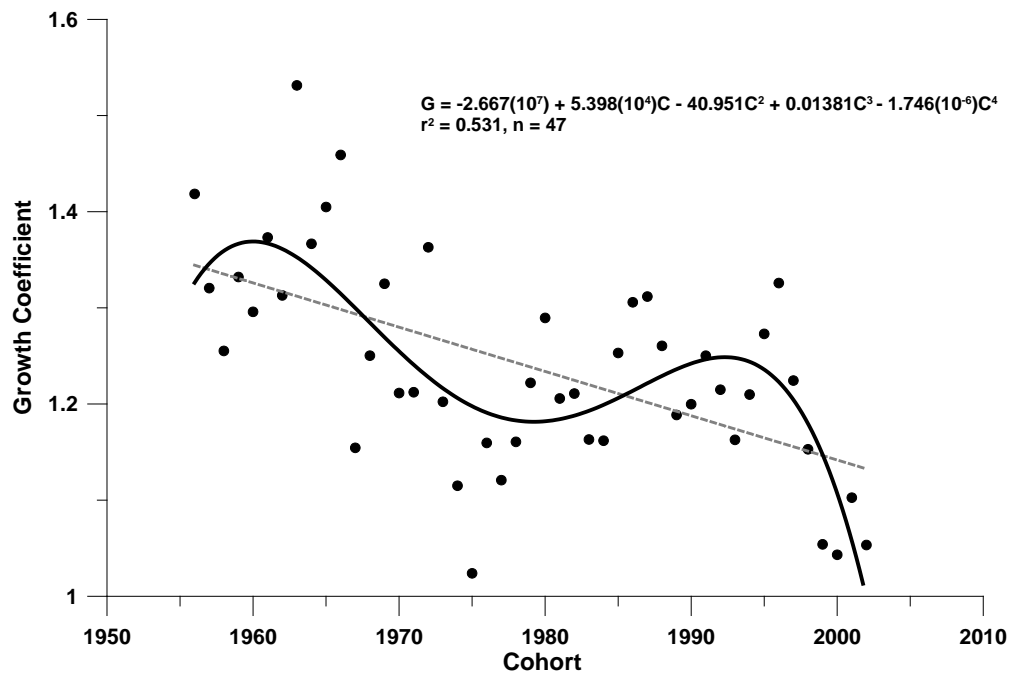
Figure 9.16. Temporal variation in North Sea plaice cohort growth coefficients. Grey dashed line shows a fitted linear trend and black line shows fitted 4th order polynomial function.

9.1.5   Fishing–induced mortality of benthos

The strong decrease of fishing effort over the last decade has reduced the level of benthic invertebrate mortality associated with fishing disturbance of seafloor habitats. However, as a result of this reduced fishing activity, plaice biomass has increased, markedly raising the natural mortality rates, associated with predation by plaice imposed on benthic invertebrates. To compare the relative changes in benthic mortality from reduced fishing mortality and increased predation by plaice, a preliminary estimate of the relative change in fishing-induced benthic mortality is made. STECF

data (http://stecf.jrc.ec.europa.eu/web/stecf/ewg1313) suggest a decrease in fishing effort expressed in Kwdays-at-sea over the past decade of approximately 50% (Figure 9.17).
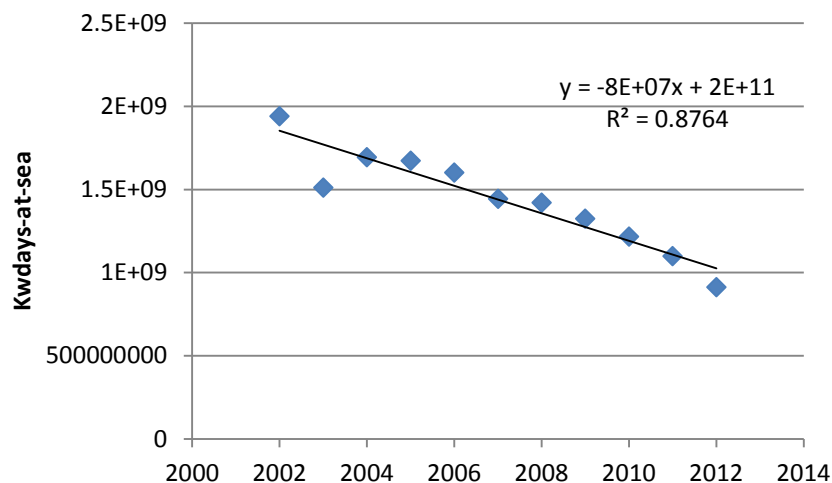
Figure 9.18. Effort in KWdays-at-sea of fisheries affecting the seafloor.

Using information on the spatial distribution of effort and how this translates into the proportion of area fished a given number of times per year (Piet and Hintzen, 2012; Figure 9.18), suggests that approximately 90% of the area is trawled less than once every year. Assuming 20% mortality per trawling event (Greenstreet *et al.*, 2007), most of the benthos mortality occurs in the lightly trawled areas. The observed estimated fishing-induced mortality is largely determined by where the reduction in effort mainly occurs. For this we applied two scenarios covering two of the possible extremes in how the spatial reduction in effort compares to the spatial patterns in fishing intensity: (1) effort is exclusively reduced in the lowest intensity areas and (2) effort is exclusively reduced in the highest intensity areas. In the first scenario, all strata with frequency <1 yr^{-1} and 60% of the frequency 1–2 yr^{-1} accounts for 50% of the effort and is hence not fished, resulting in an approximately 90% reduction of the benthos mortality (Table 9.1). In the second scenario, part of the frequency 1–2 yr^{-1} and all of the higher intensity strata accounts for 50% of the effort reduction resulting in a reduction of only 10% of fishing-induced benthos mortality.
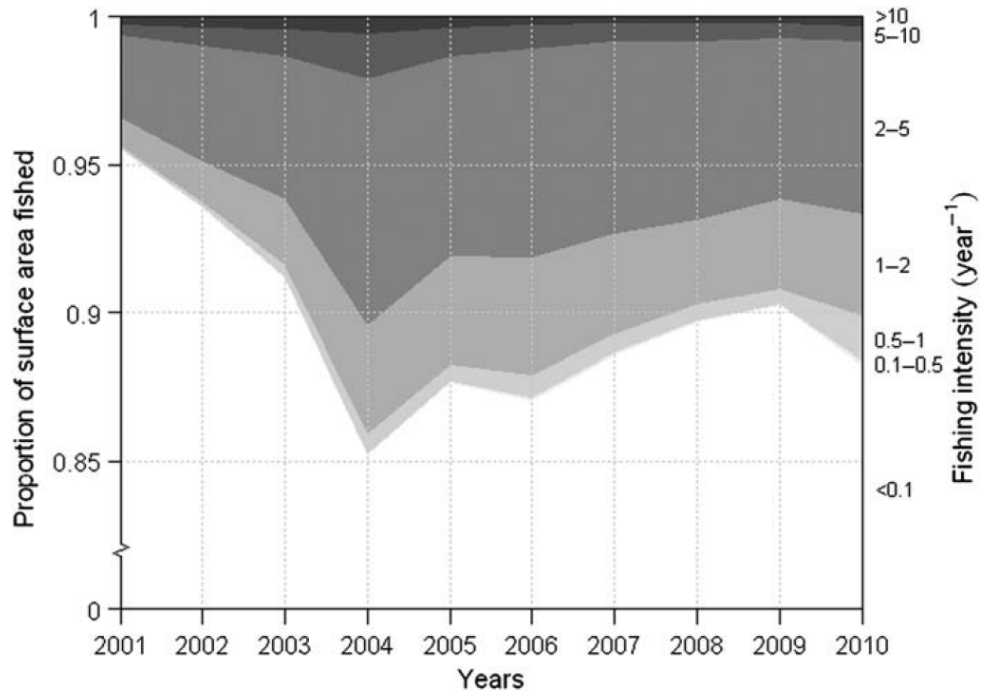
Figure 9.17. Proportion of the Dutch EEZ fished at a specific intensity over time at high resolution and based on interpolated tracks (note the y-axis break). The right y-axis indicates the intensity bins where, 0.1 means that a certain proportion of the EEZ is fished, 0.1 times per year (from (Piet and Hintzen, 2012)).

Table 9.1. Proportions of the area, estimated benthos mortality and extent of the frequency strata used depending on the two extreme scenarios of effort reduction in relation to fishing intensity.

			% OF THE AREA	
Frequency Strate (yr-1)	% Area	%Mortality	Low effort	High effort
<0.1	89.7	83.4	0	100
0.1	0.1	0.1	0	100
0.5	0.6	0.6	0	100
1	2.9	2.7	0	100
2	5.7	9.6	58.9	41.1
5	0.7	2.3	100	0
>10	0.3	1.4	100	0

In summary, mortality caused by plaice predation is assumed to have increased nine-fold the most recent years, while the fishing-induced benthos mortality has decreased by 10–90%. How these two relative changes compare in absolute terms, however, depends on how the natural mortality of benthos compares to the fishing mortality. More work is therefore required to properly assess the effects of fisheries management on the benthic community and determine if the reduced fishing-induced benthos mortality is offset by the indirect effects caused by an increased mortality of benthos by plaice and other benthivores.

9.2 Provide advice on appropriate ecosystem "process" data and underlying survey needs (WGISUR)

This section is a response to the request from WGISUR to WGECO: "It is recommended that advice be provided on how to design a survey approach to provide ecosystem "process" data, and on what "process" data would be most appropriate".

This section focuses on the survey data collection needed in the context of increasing the understanding of ecological processes. The following advice has been ordered according to its perceived priority by WGECO such that highest priority areas appear first. The prioritization does not take account of resource implications for the surveys, which may require a different priority. Most notably, the benthic sampling will require additional effort on the surveys, but is considered a key data weakness. The fish biology data are also very important, and are seen as something that could be addressed with the current fish samples taken on board.

9.2.1 Benthic ecology

The process context here is the link between benthic ecological processes and various aspects of fishing; fishing pressure, discarding, and changes in fish communities.

Proposals have already been made by WGISUR and in other fora that bottom-trawl surveys should include epibenthic sampling, ideally using 2 m beam trawls. Epibenthic sampling could also be included in conventional beam trawl surveys, where this is not already performed. WGECO support this proposal, but would also propose additional elements to underpin additional process based understanding.

It is important to develop an understanding of the link between fishing pressure, and epibenthic community structure. It is recognized it is unlikely to be possible to include a beam trawl sample for every bottom-trawl station, within the current programme, and without additional vessel time. However, an incrementally built-up map over a number of years and surveys would be a viable option. At the same time however, it would be valuable to target some sampling on an annual basis of contrasting areas of especially high and low bottom-trawling effort. Maps of fishing effort at high resolution should be available from VMS analysis. In targeting on high/low effort areas it is important to discriminate between chronically high effort areas (e.g. *Nephrops* grounds) and more recent emergent effort hot spots. Both would be valuable in understanding the processes involved. A further complication is physical disturbance by small vessels, where VMS is missing. This is probably an intractable issue for WGISUR alone, but WGECO would advise that identification of low effort areas in particular, should avoid areas that might appear low based on VMS but have high effort from smaller vessels.

The second process aspect would be links between benthic fauna and discarding in the context of the Landing Obligation in the CFP. If it is possible to identify areas of high and low discarding then repeated sampling of these locations may help elucidate the ecological importance of discards as food subsidy to the benthos.

Where it is not possible to fully characterize the benthic community, sampling and/or analysis could be focused on key indicator species. These could include sensitive species, or important scavenger species (see Section 7.2.3. for a non-exhaustive list).

For benthic sampling, stratification within meaningful ecological habitats would be much more valuable than for instance a classic IBTS rectangle approach. In determin-

ing stations for the 2 m beam trawl samples, survey operatives should also consider seabed areas that would be untrawlable with GOV but possible with this gear.

The suggestions above focus on epibenthic sampling, but WGECO believes there may be additional value in sampling infauna with grabs or similar gear. Benthic grab samples target a different component of benthic communities and may for example provide a more relevant representation of the food resource for benthivorous fish. The sampling strategy for benthic infauna may likely require a different approach than for the epifaunal survey. Replicated samples for different habitats and depth strata would make it possible to link the benthic fauna to habitat maps. Multiple samples at a station may provide better information on local variance. WGECO would recommend consulting benthic ecologists on this approach.

9.2.2 Fish biology

The key process of interest here is foodweb dynamics. Our key tool for understanding these are the variety of foodweb indicators and models that all require a range of biological data to be appropriate. Biological data of interest includes length–weight relationships, age–length relationships, maturation-at-length relationships, condition factor, and diet from stomach samples. Protocols for analyses of stomachs are available from WGSAM but could be expanded by determining benthic invertebrates to lower levels than described in the manual. Most biological data are collected for the key commercial species, but it would be equally important to have these data for, at least, the most abundant of the other fish species, e.g. common dab (*Limanda limanda*), gurnards (Triglidae), etc. Where possible, this should be developed in consultation with the appropriate foodweb modellers working in ecoregion of the surveys. Two possible approaches can be suggested:

- Single snap shot sampling possibly targeting one or more species per survey. This could be seen as providing data to populate and thereby enhance the commonly used data in e.g. FishBase. Numbers sampled should be at the same level as for the normal commercial species sampling, but an indicative sample size of at least 300 organisms could be used as a guideline.

- Annually repeated sampling on identified key species, either on the basis of their abundance, or their importance in the foodweb dynamics. Again consultation with foodweb modellers would be advised. For repeated sampling WGECO advises that this can be restricted to the species, which cumulatively make up 95% of the biomass. Where possible, annual sampling should include weight–length and maturity-at-length relationships, to allow for tracking energy allocation to somatic growth and reproduction over time. For repeated sampling, it could be possible to sample smaller numbers to obtain an idea of whether there was any substantial interannual change. If a change appears to have occurred, more sampling could then be initiated.

9.2.3 Zooplankton ecology

Zooplankton abundance, productivity, and community structure are critical factors in ecosystem modelling. They represent the top layer in NPZ models and often the bottom, boundary condition in higher trophic level foodweb models. In Europe, the main sources of zooplankton abundance, species assemblage and distribution, is the CPR. While this is valuable resource, it may have some weaknesses such as the fixed depth sampling, occasional long periods between transects, long integration period,

mesh size, and spatial coverage. The other data source would be the small number of shore stations sampling zooplankton, which provide valuable time-series, but in one location and always coastal. Zooplankton sampling on survey vessels could be valuable to supplement and compliment the CPR, and the coastal stations. A valuable approach would be to identify important locations for repeat sampling, in consultation with biological oceanographers. Ideally these should then be occupied repeatedly throughout the year to follow the annual cycle. Locations for these stations could be based on:

- ecological importance e.g. retention areas, fish spawning or nursery areas where fish larvae feed, etc.;
- gaps in the CPR coverage or located to usefully link with a coastal station;
- regular occupation by the survey vessel on most deployments regardless of core purpose.

9.2.4 Wider linking with modelling work

As with the biological work described in Section 2 above, a great deal of our process understanding is driven by ecological models. RV surveys can further contribute to the development of these models in two critical ways; filling in data gaps/weakness in the models and particularly where the models are particularly sensitive to that weakness; and collecting data for "testable hypotheses".

- Filling data gaps: One example of this would be the lack of complete fish biology data for all the species used in foodweb models discussed above. Other weakness could probably be identified by modellers and could be addressed with RV data collection. Details would likely be model and region specific and would require consultation with the appropriate modellers;
- Testable hypotheses: Again, in collaboration with modellers, this would be the development of hypotheses driven by model conclusions that could be empirically tested in the field on RV surveys. This could entail additional sampling, more detailed handling of standard sampling or in some cases changes in the protocols of sampling, provided that this did not impact on the core survey activities. These would likely be single one-off activities, and should be specifically formulated to test model driven hypotheses.

Finally WGECO also examined the table from CICES on ecosystem services (see Section 6. This work is still in progress, but may prove very useful in focusing additional survey sampling in future.

9.3 References

Fraser, H. M., Greenstreet, S. P. R., and Piet, G. J. 2007. Taking account of catchability in groundfish survey trawls: implications for estimating demersal fish biomass. ICES Journal of Marine Science, 64: 1800–1819.

Greenstreet, S, Robinson, L., Piet, G., Craeymeersch, J., Callaway, R., Reiss, H., Ehrich, S., Kröncke, I., Fraser, H., Lancaster, J., Jorgensen L. and Annelies Goffin, A. 2007. The ecological disturbance caused by fishing in the North Sea. Fisheries Research Services Collaborative Report No 04/07, 169 pages.

Piet, G.J., Hintzen, N.T. 2012. Indicators of fishing pressure and seafloor integrity. ICES Journal of Marine Science 69, 1850–1858.

10   Conclusions

WGECO specifically noted the need to enhance the development of **benthic GES indicators** through the collection of data, the development of new indicators, investigation of the importance of the benthic community to foodweb interactions, clarification of the ecosystem services provided by the benthic community and hence priority areas for indicators to be developed. Enhancing effort in these areas should also provide an increased understanding of the **effect of a landing obligation on the benthic community**, particularly scavengers, as well as an understanding of recent changes such as the increase in flatfish in the North Sea. More attention is required to determine the **effects of rebuilding predator stocks and changing the distribution of fishing mortality across species and sizes** according to their productivity (increasing "balanced fishing" efforts). Further, the **development of indicators of distribution of species** has received little attention in previous years and is recommended as a priority area of investigation for the future. Finally, WGECO considers that the identification of data needs and **recommendations for further sampling** should be an integral part of considerations for new indicators.

In accordance with these identified priority areas of study, the suggested ToRs for the next WGECO meeting are at Annex 3.

Annex 1:   List of participants

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Annex 2:   Draft Agenda

TORs referred to are given in the bottom of the document.

Tuesday April 8th

1000   Opening of the meeting

Adoption of ToR & Agenda

Overview of volunteers to work on different ToRs and overview of presentations prepared for the meeting

Initial discussion of ToR a, Continue the development of foodweb indicators and comment on the suggested foodweb indicators from WKFooWI and WGSAM

1300   Lunch

1400   Reconvene

1600   Coffee

1730   Develop Following Day Workplan, Updates from earlier in day, Wrap up

1800   Adjourn

Wednesday April 9th

0900   Initial discussion of ToR c: Consider the ecosystem consequences of "balanced fishing" regimes.

1030   Coffee

1300   Lunch

Initial discussion of ToR e: Recommend priority areas of study to determine the ecosystem consequences of landing obligations/discard bans, including survival associated with releasing fish caught

1400   Reconvene

Initial discussion of ToR f: Review knowledge of the consequences to stocks of prey fish (and other parts of the ecosystem) of restoring / maintaining stocks of predatory fish to MSY and recommend priority areas for study

1600   Coffee

Subgroups on ToRs a and d

1730   Develop Following Day Workplan, Updates from earlier in day, Wrap up

1800   Adjourn

Thursday April 10th

0900   Initial discussion of ToR d: Work towards including new research on reducing effects on the seabed and associated communities of fishing operations and gears, including ghost fishing in ecosystem advice.

Initial discussion of ToR b: Continue work on the large fish indicator, especially in waters other than the North Sea.

0900   Subgroups on ToRs b and f

1030	Coffee
	Subgroups on ToRs b and f
1300	Lunch
1400	Reconvene
	Subgroups on ToRs c and e
1600	Coffee
	Subgroups on ToRs c and e
1730	Develop Following Day Workplan, Updates from earlier in day, Wrap up
1800	Adjourn

Friday April 11th

0900	Preliminary check on WG report elements
	Revisit, as need be, ToRs a–f
1030	Coffee
1300	Lunch
1400	Reconvene
1600	Coffee
1730	Develop Following Day Workplan, Updates from earlier in day, Wrap up
1800	Adjourn

Saturday April 12th

1000	Revisit, as need be, ToRs a–f
	Subgroups
1100	Coffee
	Subgroups
1300	Lunch
1400	Reconvene
	Revisit, as need be, ToRs a–f
	Subgroups
1600	Coffee
	Subgroups
1730	Develop Following Day Workplan, Updates from earlier in day, Wrap up
1800	Adjourn

Sunday April 13th

Writing day, no plenaries

Monday April 14th

0900   Subgroup reporting

1030   Coffee

Drafting session

1300   Lunch

1400   Reconvene

Drafting session

1600   Coffee

1800   Adjourn

Tuesday April 15th

0900   Scope out next year meeting plan, schedule, ToR

1030   Coffee

Tying up loose ends

1300   Adjourn

Annex 3:   WGECO Terms of Reference for the next meeting

The **Working Group on the Ecosystem Effects of Fishing Activities** (WGECO), chaired by Anna Rindorf, Denmark, will meet in Copenhagen, Denmark xx–xx April 2015 to:

a) Develop and assess indicators of Good Environmental Status of the benthic community;

b) Develop indicators of scavengers, examine their relation to discard amounts and evaluate the potential effect of a landing obligation on the benthic ecosystem;

c) Evaluate the ecological consequences of restoring stocks to MSY levels and the degree to which fisheries are "balanced;"

d) Review, develop and assess indicators of Good Environmental Status of Distribution in the context of MSFD and management response;

e) Identify data weaknesses in the data available to address WGECO ToRs and recommend priority areas for data collection and model development.

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

Supporting Information

Priority	The current activities of this Group will 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) WGECO considers that the development and assessment of indicators of Good Environmental Status of the benthic community is a key area of development in relation to the MSFD. With this ToR, WGECO will follow and review the progress in international projects working on this topic and work towards the development of operational objectives and indicators for the benthic community as well as an improved understanding of benthic foodweb dynamics. In 2015, WGECO will speifically review the effect of trawling on benthic community composition. Term of Reference b) The implementation of a landing obligation is likely to result in major changes to the impact of fishing on particularly benthic scavenging species. The magnitude of both direct and indirect effects are unknown at present and further work is needed in order to provide advice on the direction and magnitude of change as a result of the landing obligation. WGECO considers that this ToR should be accompagnied by a similar ToR for WGBIRD. Term of Reference c) Determining the ecological consequences of restoring stocks to MSY levels and the degree to which fisheries are "balanced" ideally requires a large metadataset to ensure that analyses are based on all avialable data. To ensure that such data are available to WGECO, the group will issue a data call in january 2015 to obtain the data necessary to (i) examine time-trends in stocks that are rebuilding and dependent stocks: their prey and predators; compare these trends with predictions from qualitative analyses or LeMans or other quantitative multispecies models and (ii) establish the distribution of total catch among size classes (catch size spectrum), species (catch species dominance curves) and functional groups. The data call will request two types of data: 1. Stock assessment output for all stocks assessed in an area: F and SSB estimates tabulated in a consistent way. There is a trade-off between the number of stocks included and the length of the *time-series*. (i) needs at least 10 years, whereas (ii) might be shorter especially if it allows to include more discard estimates. (ii) may be more a snapshot that could be compared across areas. 2. (ii) may also be adressed based on catch data = landings + discard estimates. Landings are available from public databases; discard estimates for TAC-species are going to be available in the discard atlases being prepared as input to the regional discard management plans; issue is discard estimates for non-TAC species. Term of Reference d) Distributional indicators have obtained little attention in ICES since the end of WGFE. WGECO considers this an

important area of research in connection with the MSFD and proposes that the work should be continued in cooperation with e.g. SGSPATIAL. The work in 2015 will include reviewing of existing indicators, intial identification of development needs and evaluation of the possible repsonse of distributional indicators to management.

Term of Reference e)

WGECO considers that it is often of use to other grooups to receive clear guidelines on priority areas of data collection or model development. Therefore, this should be a standing ToR of WGECO with resulting advice to other groups given based on the progress made in the particular year.

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 obvious 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	

Request to other groups

REQUEST	GROUP	EXPLANATION
Data or other information on spatial and temporal patterns in benthic communities	BEWG	In 2015, WGECO will specifically work with the development of indicators of GES of benthic communities. To this aim, having data on either spatial distribution or temporal development at specific stations of benthos is crucial. Hence, any data of this kind or expertise on this area that BEWG could provide would be of great value to WGECO.

Annex 4:   Technical Review of Indicators for MSFD Descriptor 4

Summary of reviews of WKFooWI and WGECO

This document is a synthesis of the independent reviews of the work of WKFooWI and the work of WGECO in readiness for the drafting of ICES advice. WGECO also commented on the work of WKFooWI, and this is included in this synthesis.

Overall summary

The reviewers appear content that the indices were evaluated appropriately and using suitable criteria. There was some criticism of the inadequate descriptions of each index. One reviewer felt that the definitions of structure, function and resilience need clarification and that indices were perhaps inappropriately classified. The issue of indices for management action and indices for surveillance of change (no direct pressure to state relationship, e.g. zooplankton biomass index) was discussed and needs to be highlighted. This should be clarified for each of the five in the suite of 5 proposed indices. The suite of 5 was broadly accepted by the reviewers although one reviewer proposed that two other types of indices were missing (structural foodweb index for uni-cellular organisms and a topological index (who eats who). There was criticism of the roadmap (with an alternative roadmap provided), especially for the development of targets or thresholds. There was a request to make sure that the advice links through to the previous ICES advice on DCF time-series for the MSFD.

Little extra insight was provided about the LFI work by the reviewers. Considering that the LFI is included in the MSFD legislation, and appears to now be moved from D4 to D1 by the scientific community, neither WKFooWI nor WGECO concisely addressed what the MSFD should do with the LFI.

1. Foodweb indicator development carried out at WKFooWI

1.1 WGECO comment

WGECO noted that WKFooWI recognized the following key elements of a process for choosing indicators:

- The need to have a suite of indicators, and not just the "one" indicator;
- The need to have clear criteria for selecting indicators;
- The need to have clear objectives for why indicators shall be developed and used;
- The need to have clear venues for evaluating, vetting and referencing indicators;
- The need to have clear "clients" who will use the indicators and are asking for them.

In addition, indicators should be sensitive, have a basis in theory and be measurable. The evaluation criteria were availability of data, quality of underlying data, conceptual/theoretical basis, communication and manageable. WKFooWI distinguished the attributes of a foodweb characterized by an indicator (structure, function, resilience) and what they called a foodweb indicator class (energy flow, network, canary, diversity, size, aggregate). It is also important to consider functional groups (phytoplankton, zooplankton, benthos, cephalopods, fish, birds, mammals, reptiles). WGECO

then provide a table of which potential indicators were primarily associated with which foodweb attributes (WGECO Table 3.1). WGECO agreed that the evaluation of the indicators was carried out following the accepted methods developed by WGECO and WGBIODIV.

WGECO made the following observations about the five indicators recommended by WKFooWI as the initial suite of indicators.

INDICATOR	RATIONALE	WGECO OBSERVATION
Guild level biomass (and production)	Structural attributes of foodwebs, and can also serve as a proxy for functioning. Improved specification of MSFD D4 indicator, Production per unit biomass 4.1.1 as well the D4 indicator abundance within range 4.3.1.	This would definitely be useful as a surveillance indicator[1] for the state of the foodweb and the relative stability of its major components. As an operational indicator, it may be difficult to manage, particularly through fishery measures. Given our current state of knowledge, it may also be difficult to set specific targets for the biomass of particular guilds. If management were possible, it may well end up with a focus on particular species within a guild where fisheries measures might be more effective.
Primary Production Required to sustain a fishery	The functioning attribute of foodwebs. Improved specification of D4 indicator, Production per unit biomass 4.1.1.	This would appear to be primarily useful as a surveillance indicator[1]. It is difficult to see how specific management could be exerted. If trophic level of specific groups is not constant, the indicator requires persistent sampling of diet composition. It requires context setting and can be difficult to communicate.
Seabird (charismatic megafauna) productivity	The structural attribute of a foodweb, and may be able to serve as a proxy for resilience or functioning. Improved specification of D4 indicator, Production per unit biomass 4.1.1	These indicators have already been well documented and used in a range of contexts, and can be considered as operational and suitable for management. In the full version of the WKFooWI report, seabird productivity is directly cited as expressing the "abundance" of forage fish, while it actually probably reflects the "availability" of these fish. These indicators are undoubtedly valuable in themselves, but maybe questionable in terms of "integrating" the foodweb below them.
Zooplankton spatial distribution and total biomass	Both structural and functional attributes of foodwebs.	This would be a surveillance indicator[1], for general ecosystem health and productivity–but would not be manageable.
Integrated indicators (mean TL, mean size)	Both structural and resilience attributes of foodwebs.	Again, this is a good surveillance indicator. Like guild level biomass, it may be potentially subject to management that focuses on individual components of the community

WGECO then stated that the most valuable indicators are those which are operational and appropriate to direct management via a pressure–state relationship. There are also surveillance indicators that are indicators that quantify neither pressures nor directly affected attributes, but are nevertheless needed for an informed assessment and management of foodwebs. A key feature of surveillance indicators is that they are

unlikely to respond unequivocally to management or support target setting. They operate more to provide warning of changes that may impact on our ability to achieve targets in other indicators (e.g. zooplankton biomass).

WGECO then suggest caution when using "fish" dominated approaches, or approaches that assume foodwebs based on "adult only" diets.

1.2 Nik Probst

Why did WKFooWI simplify the evaluation criteria previously used by WGBIODIV? However the simplification appeared appropriate. More descriptions of the indicators would have been beneficial. The following work is required to make the indicators operational by 2018.

a) Specification of indicator metrics.

b) Gathering of relevant data.

c) Analysis of pressure–state relationship.

d) Development of indicator targets.

e) Constant updating and reassessment (also of targets).

Why were so many indicators scored highly for the criterion "management thresholds (targets) estimable". Why for indicators such as "biomass of trophic guilds" this criterion scored also highly. Was the thinking that healthy or good ecosystems consist of large, predatory fish (gadoids for the best) without scavengers and lower trophic groups. Whether this is ubiquitously the case, can be questioned. In fact, exploited systems may be modified, but also healthy and stable.

Also the assumption by WKFooWI that the best indicators are based on observed (empirical) rather than modelled data was supported.

1.3 Simon Jennings

The work of WKFooWI was much more focused than that of WGECO and will be easier to turn into advice. WKFooWI were clear that they were aiming for pragmatic approaches to identify, use and continue to develop FooWI. The analysis was complete to the extent possible. The shortlist of indicators provide a suitable focus going forward, provided ICES can move quickly towards developing the technical specifications for these general classes of indicator.

WGECO commented that several of the short-list of indicators proposed by WKFoo-WI are surveillance indicators. Given there is no technical description of the indicators this is a reasonable analysis based on current understanding of pressure–state links, but further selection and technical development of these indicators could tailor them to respond to impacts we can actually manage.

The focus on the development of a roadmap was limited (question c) and plans for moving towards future specification and implementation of D4 indicators are not clear. The WKFooWI report does define a process for selecting and developing D4 indicators and then applies it, and these are two important first steps in a longer process that might be described in a 'roadmap'. The advice could therefore show that two steps in a mapped process were complete, but would need to articulate the other steps, perhaps drawing on experience with D3, for which planning is more advanced than for the rest of the interrelated D1, D3, D4, D6 group. In the 'Roadmap' section of the WKFooWI report it is perhaps optimistic to brigade the short-list as suggested

FooWI for current use, as I do not see evidence of technical underpinnings needed to use them right away in the MSFD context; although some have been the subject of research papers etc. and some components of these indicators are already available/ used in other contexts.

Possible steps for a roadmap that includes the steps already presented would be:

a) define criteria for selection of broad indicator classes (done WKFooWI and others);

b) make selection of priority broad indicator classes based on criteria and map to EC(2010) (done WKFooWI);

c) develop technical specification of indicators within the selected broad classes at Regional scales, taking account of contributions of existing indicators (D1, D4, D6) and available data;

d) screen refined indicators against criteria (strongly engaging RCS and representatives MS);

e) write up technical specifications of indicators that pass screening in clear accessible format, provide 'toolkit' for RSC and MS to generate and report indicators that pass screening.

With regards to the selected initial suite of indicators:

Guild level biomass (and production): If the initial aspiration is not to be comprehensive then significant initial progress will be made by drawing on data and indicators for other descriptors. This approach would also solve the challenge of identifying indicators that respond to management measures. For fishes, guilds could be based on the sum of biomass or production from groups of assessed stocks, especially when these cover a large proportion of biomass regionally. If large proportions of biomass in functionally important guilds are not covered at present in some regions then additional population assessments might be conducted to fulfil the aim of developing indicators for the guild (e.g. previous (2013) advice that assessments of all forage fish species that account for >5% of total fish biomass, or that are important in the diet of dependent species (especially when these are protected species)). These may support D3 as well. For higher predators (e.g. mammals and birds) estimates of abundance and production that would also fulfil the needs of D1 could be used and presented in aggregate form to support D4. Primary production from remote sensing already well supported by work of JRC, and this relates to the second of the short-list of indicators as well. However, the issue with moving away from species sensitive to the various types of mortality imposed by people (or the few cases where there is a well-established indirect response) will be that there is no identifiable management measure for MS to put in place. For this reason, and given criteria, I suggest the strength of pressure-state links may be used in the roadmap to help prioritize the work on guilds.

Primary production required to sustain a fishery: Since landings data are readily available at appropriate scales this indicator can be calculated with information on trophic level at size of the fished species, primary production and assumed transfer efficiency. No limits/ targets are clearly justifiable at the moment so far as I am aware, but the value of the indicator would respond to management if you wanted it to. Lots of likely controversy surrounding trophic level and transfer efficiency as assumptions here have a big effect on outcomes. However, cheap to calculate and applies to all regions.

Seabird (charismatic megafauna) productivity: Well developed and could also serve D1 and input to the guild analysis above.

Zooplankton size biomass index: If zooplankton assessment of some form were attempted this would also support the guild analysis above.

Integrated trophic indicators (mean TL, mean size): I assume this is where you assume LFI or a proxy is retained, maybe worth stating explicitly to link to the other ongoing and reported work. The two examples used in your title for this indicator are less understood and perform less effectively in most case studies the slope of size spectra, note also WGECO analysis in the reviewed section on large fish and trophic level (and concluded that the strength of connection was variable) so need to check consistency of message in material presented.

1.4 Benjamin Planque

The workshop report provides a clear answer to the request by the EU to ICES on the development of criteria and potentially useful indicators to address the relationships within the foodweb. Thus the objectives of the workshop, i.e. to produce a short list of foodweb indicators for the EU-MSFD and a defined process for selecting these indicators, were met. The methods used to evaluate the criteria were valid and conformed to acceptable norms. WKFooWI also accounted for its own internal bias.

WKFooWI choose to partition the indicators into three main groups 1) functional indicators linked to energy flows, 2) functional indicators linked to ecosystem resilience and 3) structural indicators linked to diversity and 'canary' species. This partition of the indicators was not so easy to follow and that several indicators could easily have been moved to another category. The preferred approach would be to consider:

- Foodwebs can be defined as networks in which nodes are trophospecies (which can be individual taxa, guilds, size-based groups of individuals, etc.) and connections between nodes are trophic flows (often expressed in mass, carbon or energy).

- A foodweb structure can often be described by its topology (i.e. the listing of trophospecies and trophic flows) eventually complemented by quantitative estimates of biomasses.

- The dynamics within the foodweb is best described by quantification of the trophic flows, how they vary over time and how they affect trophospecies biomass. In addition, reconfiguration of the foodweb topology may occur (by extinction or colonization).

- A pragmatic approach to the description of resilience in foodweb is provided in Levin and Lubchenco (2008) who identify three important qualities that confer resilience to networks: diversity, redundancy and modularity. This paper should have been referenced.

It is suggested to re- group the general categories and re-adopt the ones outlined above: structure, dynamics and resilience. This would not affect scoring and evaluation of individual indicators.

A primary focus is made on pressure-response and the establishment of rigorous thresholds for indicators. In many cases however, multiple synergistic pressures may prevent from establishing easy pressure-response relationships and associated thresholds. A balanced view between the use of indicators against thresholds and the

use of trend-based assessment using indicators without threshold might be more appropriate.

The section on descriptions of the indices provides the rational for including individual indicators in the evaluation/selection process. However, this seems to have been written by many hands and the result is uneven. Some sections provide measurement/calculation methods, some provide guideline for interpretation, some provide indication of applicability for management, but few provide all of the above. A standardization of these sections would be helpful and useful.

There are two types of indicator missing from the list:

1) On the lower end of the pelagic foodweb lie unicellular organisms which can be autotrophs, heterotrophs or mixotrophs and belong to various taxonomic groups (e.g. bacteria, protozoans, diatoms, …). This part of the foodweb is believed to be particularly sensitive to warming and acidification of the ocean with responses that might likely percolates to higher trophic levels. These were not included as indicators changes in structure of dynamics in the lower part of foodwebs.

2) One of the simplest ways to describe a foodweb is a topological description (i.e. who eats whom). Surprisingly, no indicators of foodweb topology are presented.

Why did none of the five include an indicator for resilience?

2. LFI analysis carried out by WGECO

Overview of currently published regional LFIs and ongoing work

Area	LFI DEVELOPMENT STAGE	TIME–SERIES	SPECIFIC TRESHOLD DEFINED	SPECIFIC REFERENCE LEVEL
North Sea	Completed[1]	Yes	Yes	Yes
Celtic Sea	Completed[2]	Yes	Yes	Yes
Southern Bay of Biscay	Completed[3]	Yes	Yes	Yes
Central-Southern Tyrrhenian Sea	Ongoing[4]	Yes	No	No
Baltic Sea	Ongoing[5]	Yes	Yes	No
Poland EEZ	Completed[6]	Yes	Yes	Yes
Kattegat North	Ongoing[7]	Yes	No	No
Kattegat South	Ongoing[7]	Yes	No	No
The Sound	Ongoing[7]	Yes	No	No
Gulf of Cádiz	Ongoing[8]	No	No	No

2.1 Nik Probst

Nik reviewed Chapter 3 of the WGECO report.

2.2 Simon Jennings

The WGECO report contains extensive new work on the LFI and, when edited, this will therefore fulfil the DGENV request (question a). Since the ToR for WGECO was simply to continue working on LFI the work is necessarily not complete. I agree with most of the scientific conclusions but they are not strongly focused on application in the management system (if anything previous WGECO reports have been stronger in this regard). However, the work remains predominantly exploratory and descriptive, as it has for a number of years, and still has some way to go in terms of reaching maturity (agreed specifications and code for calculation that could be shared among MS and passed to other EG for example, good understanding of responses to alternate management actions).

WGECO did fulfil their ToR to extend the work to areas outside the North Sea. Although DGENV simply ask for ICES to continue working on the LFI, this work has been going on for several years now and I hope you can craft the advice to show clear direction in the new work being done and perhaps encourage more specific goal oriented requests that can then be passed to the relevant EG in future. My concern is that the group working on this topic are very good at continuing work, but also need to develop the work in a way that can be used by MS that may ultimately implement these methods (either inside or outside ICES fora).

2.3 Benjamin Planque

No comments with regards to the LFI work.