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11–18 April 2012

Copenhagen, Denmark

ICES
International Council for the Exploration of the Sea
CIEM
Conseil International pour l'Exploration de la Mer

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Executive Summary

The 2012 meeting of WGECO was held at the ICES HQ in Copenhagen, Denmark from the 11–18 April 2012. The meeting was attended by 27 delegates from 13 countries, and was chaired by Dave Reid (Ireland). The WG addressed seven terms of reference.

a) MSFD indicator and target evaluation. Request from ACOM;
b) DCF Indicators on fishing pressures/impacts and possible developments/ improvements in these. Request from ACOM;
c) Evaluate approaches and propose elements for inclusion for Ecosystem Impact Assessment of bottom fishing. Request from NEAFC;
d) Indicators of foodweb condition in reference to MSFD Indicator 4;
e) LFI development;
f) Review of the state-of-the-art in understanding the combined effects of multiple ecosystem pressures, including advice for future research;
g) Trade-off in biodiversity conservation and sustainable use.

The first two ToR were based on requests from ACOM in the context of ICES advice on the MSFD and CFP reform processes. The third was based on an advice request by NEAFC. The remaining four were based on ongoing work of WGECO designed to inform the process of including ecological indicators in management, and particularly in the context of the MSFD.

The first ToR on developing criteria for the evaluation of indicators and targets chosen for GES by various member states turned out to be very difficult to complete, and the report given here should only be considered as an indication of work in progress. Essentially, the work on criteria for indicators condensed and updated a number of other approaches already in the public domain and tried to take the best of these and where necessary supplement them. Criteria for evaluating target levels were essentially taken as refinement of the criteria for indicators, although it was recognized that these could be added to. A number of approaches to analysing multiple indicators and/or surveys were also explored. Again, this was work in progress and will be continued next year, and was reported as such in **Chapter 3.**

The DCF has stipulated a number of indicators for collection and analysis over recent years. The WG examined a number of these including:

- Conservation status of fish species;
- Proportion of large fish;
- Mean maximum length of fishes;
- Size at maturation of exploited fish species;
- Fishing pressure based on VMS;
- Discarding rates of commercially exploited species;
- Fuel efficiency of fish capture.

Each indicator was evaluated in terms of the calculations stipulated, and outcomes for a number of directed case studies. A number of suggestions for changes and improvement were outlined in **Chapter 4.**

The third ToR considered ecological risk assessment and its particular application to deep-sea fishing in the NEAFC region. Initially the WG reviewed a wide range of ecological risk assessment methodologies and evaluated their strengths and weakness. This included a number of case studies carried out for each example. Based on this study a series of recommendations were developed for NEAFC, and these are reported in **Chapter 5.**

The MSFD includes at descriptor 4 an area where the decision document considers that there is a considerable need for further research. The initial work examined any gaps and shortcomings of current MSFD Foodweb Indicators. The broad conclusion was that ideally one would want indicators of foodweb *function* but that for the most part; data available was restricted to foodweb *structure*. The role of models to elucidate the link between structures and functions was examined, and the possible use of fluxes as foodweb functions. The work then went on to identify a series of attributes of foodweb structure that should be maintained, and proposed approaches to work with these through structural metrics, reported in **Chapter 6.**

The Large Fish Indicator (LFI) has been a focus of work for WGECO for some years and this was continued in 2012. In particular we looked at tuning the indicator for the Baltic Sea and the Gulf of Cadiz. In addition we explored the potential effects of interaction between different sampling gears and fish communities. Finally, we considered the application of the LFI in MSFD subregions which are sampled by several independent fisheries surveys. This work is reported in **Chapter 7.**

Cumulative impacts were investigated to investigate the occurrence of additive, synergistic and antagonistic impacts of pressures and climatic drivers. Potential interference between multiple pressures is likely to be widespread; and the first attempts at mapping multiple pressures suggest that this potential overlap occurs in many parts of the marine environment. The few experimental studies available, and some worked examples, suggest that, more often than not, interactions between multiple pressures are not simple; with both synergistic and antagonistic effects happening. The development of integrated models will be necessary to better frame research needs and support management; as well as for complex statistical models, this might require increased computational power. There is also a need for field studies of the most important interactions at various scales. Integrated observation systems are going to be increasingly needed, with a wide variety of physical, biological, ecological and human observations collected on potentially large-scales. Active adaptive management *sensu* Walters (1986) may be very useful to investigate and manage cumulative impacts. This work is reported in **Chapter 8.**

Trade-offs between biological conservation and social and economic targets were the subject of the final ToR. Our review first looked at a range of international conventions (CBD, MSFD, CFP) and what was said about societal and economic targets and trade-offs. We then examined what social and economic indicators were available for use, or at least potentially. The next step was examination of a number of possible methodologies for examining such trade-offs, and finally a number of case studies where attempts to quantify what is involved in such trade-offs was examined. This work is reported in **Chapter 9**.

1 Opening of the meeting

The meeting was opened at 10.00 am on 11 April and adjourned on 18 April 2012. The meeting was chaired by David Reid, Ireland, and attended by 27 participants from 13 different countries. Two of the participants contributed by correspondence. A full participants list is found at Annex 1.

2 Adoption of the agenda

The agenda was considered. The draft agenda is found below.

1000 Wednesday 11 April

Plenary

Introductions

Presentation on using ICES SharePoint/Printer and other services

Overview of meeting work plan. **Dave Reid**

Presentation on WGECO approach to **ToR a**: MSFD indicator and target evaluation. **Simon Greenstreet & Leonie Robinson/Tony Knights**

Presentation on WGECO approach to **ToR b**: DCF Indicators. **Sam Shephard & Gerjan Piet**

Presentation on WGECO approach to **ToR c**: Ecosystem Impact Assessment of bottom fishing. **Jochen Depstele, Heino Fock & Ellen Kenchington**

Presentation on WGECO approach to **ToR d**: Indicators of foodweb condition in reference to MSFD Indicator 4. **Axel Rossberg & Fatima Borges**

Presentation on WGECO approach to **ToR e**: LFI development. **Daniel Oesterwind & Elena Guijarro.**

Presentation on WGECO approach to new **ToR f**: Cumulative impacts from multiple pressures – **Marie-Joelle Rochet & Anna Rindorf**

Presentation on WGECO approach to new **ToR g**: Trade-off in biodiversity conservation and sustainable use **Jeremy Collie & Vanessa Steltzenmüller**

Getting the show on the road

Allocation of people to ToR

Discussion groups for ToRs a–f;

Uploading material to SharePoint, etc.

0900–1000 Thursday 12 April

Meeting of ToR leaders to inform each other of direction each group is taking

1100–1200 Plenary for any emerging issues

0900 Friday 13 April

Discussion groups for all ToRs

***** Meeting to follow a format of break-out group and plenary discussion as required with times to be posted daily based on progress *****

Weekend: WGECO works through both Saturday and Sunday with a later start on Saturday and a late day plenary on Sunday.

Tuesday 17 April

The last plenary session will be scheduled for the afternoon. Remaining time will be spent tidying up the report, finalizing references, etc. Each ToR group should identify at least one member who will be present Tuesday afternoon to do this. There will be no formal meeting on the Wednesday as I anticipate a lot of early leavers!!!!!

3 ToRa) MSFD indicator and target evaluation

Term of Reference a) MSFD indicator and target evaluation:

> To develop, test and report on (a) criteria and a process for evaluating the scientific soundness and feasibility of national proposals for indicators and targets used to support the achievement of Good Environmental Status and (b) approaches for combining information provided by indicators and targets into an assessment of status. The focus should be on descriptors 1, 3, 4 and 6, but to the extent possible the criteria and process should be general for all the Descriptors and their associated indicators and targets. (ACOM request).

The response to the two parts of the ToR (a & b) are presented separately below.

3.1 Criteria and a process for evaluating the scientific soundness and feasibility of national proposals for indicators and targets used to support the achievement of Good Environmental Status

3.1.1 Introduction

The term of reference as set was considered as exceptionally broad, and WGECO agreed that this could not be fully achieved in one working group meeting. On that basis, the text below should be considered as a work in progress, rather than in any way a completed product. The initial approach was focused on developing a process for evaluating scientific soundness of proposals for GES indicators and targets. Evaluating targets was considered as more complex than evaluating indicators, and this latter work is more developed. There was no scope for actually testing such criteria as none have yet been finally proposed. The second part of the ToR was addressed based on work in progress in several MS.

It should be emphasized that this section represents a work in progress, and that no consensus on recommendations was reached by the WG. WGECO would wish to revisit this ToR next year. In the meantime, WGECO would welcome responses on the text from interested parties, and suggestions for improvement.

3.1.2 Criteria to evaluate metrics and indicators

WGECO considered existing literature on "criteria to assess the suitability of state indicators". Kershner *et al*., (2011) reviewed several studies (Box 1) and recommended a set of 19 criteria to assess the suitability of different marine ecological state indicators. Building on the work of Kershner *et al*. and taking account of the seven criteria previously proposed by ICES (2001), WGECO has derived a set of criteria that could be used to assess the usefulness of metrics proposed by member states to populate the indicators stipulated by the EC (2010) under the Marine Strategy Framework Directive (MSFD).

Box 1. List of studies cited by Kershner *et al.*, 2011 to derive "criteria of indicator suitability"
1. Doren RF, Trexler JC, Gottlieb AD, Harwell MC (2009) Ecological indicators for system-wide assessment of the greater everglades ecosystem restoration program. Ecological Indicators 9: S2–S16. 2. Harwell MA, Myers V, Young T, Bartuska A, Gassman N, *et al*. (1999) A framework for an ecosystem integrity report card. Bioscience 49: 543–556. 3. Jackson LE, Kurtz J, Fisher WS (2000) Evaluation guidelines for ecological indicators. EPA/620/R-99/005 US Environmental Protection Agency, Office of Research and Development, Research Triangle Park, NC. 107 p. 4. Jennings S (2005) Indicators to support an ecosystem approach to fisheries. Fish and Fisheries 6: 212–232. 5. Jorgensen SE, Costanza R, Xu FL (2005) Handbook of ecological indicators for assessment of ecosystem health. Boca Raton, FL, USA: CRC Press. 6. Kurtz JC, Jackson LE, Fisher WS (2001) Strategies for evaluating indicators based on guidelines from the Environmental Protection Agency's Office of Research and Development. Ecological Indicators 1: 49–60. 7. Landres PB, Verner J, Thomas JW (1988) Ecological uses of vertebrate indicator species – a critique. Conservation Biology 2: 316–328. 8. Niemeijer D, de Groot RS (2008) A conceptual framework for selecting environmental indicator sets. Ecological Indicators 8: 14–25. 9. Noss RF (1990) Indicators for monitoring biodiversity: a hierarchical approach. Conservation Biology 4: 355–364. 10. O'Connor JS, Dewling RT (1986) Indices of marine degradation: their utility. Environmental Management 10: 335–343. 11. O'Neill SM, Bravo CF, Collier TK (2008) Environmental Indicators for the Puget Sound Partnership: A Regional Effort to Select Provisional Indicators (Phase 1). Summary Report. Seattle, WA: National Oceanic and Atmospheric Administration. 12. Rice J (2003) Environmental health indicators. Ocean & Coastal Management 46: 235–259. 13. Rice JC, Rochet M-J (2005). A framework for selecting a suite of indicators for fisheries management. ICES J Mar Sci 62: 516–527.

ICES WGBIODIV (2012) reported progress by Member States prior to February 2012, where information was available, on the development of targets and indicators. Since February, most Member States have moved into a consultation process, for example the UK started its consultation process on 27 March 2012. No Member State has yet reported formally the indicators and targets they intend to put forward to meet the July 2012 MSFD obligation. Most Member States were represented at an OSPAR ICG-COBAM workshop in Amsterdam in November 2011. This workshop was intended to promote coordination between Member States in their efforts to implement the MSFD. The report of this meeting lists "common indicators" that might be used by Member States for achieving GES in shared MSFD subregions. In addition to deriving generic criteria to assess indicator suitability from first principles, WGECO examined the list of potential "common" indicators proposed by OSPAR for use by different Member States who share regional seas, to determine whether their current proposals indicated the need for further additional "criteria of indicator suitability" quite specific to the MSFD implementation process. Table 1 lists the resulting 18 criteria of indicator suitability.

Table 1. List of criteria by which to assess the suitability of metrics used to populate indicators for Descriptors 1 "Biodiversity", 4 "Foodwebs" and 6 "Seafloor integrity".

Number	Category	Characteristic	Criterion	Example
1	Quality of underlying data	Metrics should be tangible	Metrics used to populate indicators should ideally be easily and accurately measured or determined using technically feasible and quality assured methods. Quantitative or qualitative measurements may be used (see next row)	The length of fish can be easily and accurately measured, as can its weight. These indicators of body condition are easily and accurately determinable.
2		Quantitative/qualitative	Quantitative measurements are preferred over qualitative, categorical measurements, which in turn are preferred over expert opinions and professional judgments.	A quantitative indicator, such as fish condition determine as a function of weight divided by length, is preferable to a qualitative indicator such as a categorical index of "fatness"
3		Existing and ongoing data	Indicators should be supported by existing data to facilitate evaluation of current status, to set reference baselines according to status at particular times in the past. Monitoring programmes must be ongoing so that future trends can be assessed. Data should be collected on multiple, sequential occasions using consistent protocols, which account for spatial and temporal heterogeneity.	Determination of the North Sea LFI and EcoQO target value of 0.3, based on existing first quarter international bottom-trawl survey (IBTS) data, and using future survey data to evaluate progress towards achieving the EcoQO.
4		Relevant spatial coverage	Data should be derived from a large proportion of the area to which the metric will apply.	The North Sea IBTS covers most of OSPAR region II, allowing reporting of progress towards the EcoQO for the LFI. But in OSPAR region III no single survey covers the entire region, so currently only a Celtic Sea LFI and EcoQO have been defined.
5	Responsiveness	Responds to change in ecosystem attribute	If the attributes of the ecosystem change in a particular way, then the indicator should respond predictably and sensitively, reflecting that change to a predefined level of accuracy.	When population size of a demersal fish species declines by 50% in a regional sea, then groundfish survey indices of that species abundance in the region should reduce by a similar order of magnitude.
6		Responds to change in specific pressures	The indicator should respond sensitively to particular changes in a pressure (e.g. fishing mortality, habitat destruction). The response should be unambiguous and in a predictable direction, based on theoretical or empirical knowledge, thus reflecting the effect of change in pressure on the ecosystem component in question. When assessing sensitivity of metric to changes in pressure, natural time-lags in the biological processes involved need to be taken into account.	A rapid increase in chlorophyll concentration can occur in response to specific nitrate enrichment events.
7		Signal-to-noise ratio	Metrics should be measurable accurately enough that any change or trend in a given metric is greater than the variance in its measurement	Development of the LFI in both the North Sea and the Celtic Sea has involved a process of species selection and "large fish" length threshold selection specifically to increase the metric's signal-to-noise ratio.

Number	Category	Characteristic	Criterion	Example
8	Management	Relevant to management objectives	Indicator provides information related to specific management objectives, thereby supporting targets setting such that indicator tendency towards target values, or trend directions, represent real progress towards high level management goals, such as achieving GES.	Kittiwake breeding success remains at 0.6 or more chicks per pair implying that pelagic fish prey food resources to top predators are sufficient, and not depleted by industrial fishing activities.
		And to MSFD indicators in particular	Metrics should match the MSFD indicator's stated purpose so as to avoid confusion when assessing progress towards GES at the criterion level. This might be considered an MSFD-specific extension of criterion 7. 11	The OSPAR document lists "Mean maximum length (MML) of demersal fish and elasmobranchs" as a metric to populate indicator 1.6.1 "Condition of the typical species and communities". Indicator 1.6.1 is a "habitats" level indicator. Indicators derived from survey data of "mobile" species such as fish should be used to populate "species" or "ecosystem" level indicators. It may be more appropriate to use this metric to populate indicator 1.3.1 "Population demographic characteristics", a "species" level indicator. However, the MML as defined by CEC (2008) is a "community level" metric. Since it is not a species-specific indicator, it might be more appropriate to use this metric to populate the "ecosystem" level indicator 1.7.1.
9		Relevant to management measures	Indicator links directly to management response.	Commercial fish stocks should be fished at maximum sustainable yield (MSY), so if fishing mortality (f) > fMSY, then fishing activity should be reduced.
10		Comprehensible	Indicators should be easily understandable by policy-makers and other non-scientists alike, and clear to communicate.	Variation in size is a concept grasped by every, so variation in metrics of mean length would be a metric easily understood by any lay-person.
11		Established indicator	Indicators already perceived by policy-makers and non-scientists as reliable and meaningful should be preferred over novel indicators that perform the same role.	Spawning-stock biomass is an indicator of the state of particular commercial species that has a long established track record within the policy arena.
12		Cost-effectiveness	Sampling, measuring, processing, analysing indicator data, and reporting assessment outcomes, should make effective use of limited financial resources.	Undertaking multiple monitoring tasks on single research vessel surveys is potentially better use of resources, than using separate trips to address different monitoring needs individually.
13		Early warning	Indicators that signal potential future change in an ecosystem attribute before actual "harm" is indicated by other MSFD indicators should be preferred. These could allow "preventive" management, which could be less costly that "restorative" management.	Early management action in response to herring and cod recruitment indicators could have prevented stock collapses, and been much less costly from a socio-economic perspective than the subsequent recovery management measures that were needed.

Number	Category	Characteristic	Criterion	Example
14	Conceptual	Theoretically sound	Scientific, peer-reviewed findings should underpin the assertion that the metric provides a true representation of variation in the ecosystem attribute in question	There is a sound body of peer-reviewed evidence to support the contention that age structure in fish populations tends towards younger-aged and size structure tends towards smaller-sized fish in heavily exploited populations.
15		Indicator suites[1]	Sets of indicators should be selected to avoid redundancy, increase the complementary of the information provided. Different indicators within the set should reflect change in different ecosystem attributes.	The use of indicators of species richness, species evenness, abundance, size composition, life-history trait composition to cover all major aspects of potential structural and functional change in fish communities (Bundy *et al*., 2010; Greenstreet *et al*., 2012).
16		State or pressure	It should be unambiguous whether the metric is being used as an indicator of "state" or "pressure", so as to avoid confusion in respect of targets.	The OSPAR document lists "Bycatch rates of Chondrichthyes " as a metric to populate indicator 1.2.1 "Population abundance". If bycatch were being used as a proxy of the abundance of chondrichthyes species, then the target would be "Increase the bycatch", since chondrichthyes species are generally considered to be sensitive species whose populations have been depleted by fishing mortality. The stated target "bycatch reduced in stipulated number of Chondrichthyes species", however, implies that the metrics is being applied as a pressure indicator.
17		Metrics relevance to MSFD indicator	Metrics should match the MSFD indicator's stated purpose so as to avoid confusion when assessing progress towards GES at the criterion level. This might be considered an MSFD-specific extension of criterion 7.	The OSPAR document lists "Mean maximum length (MML) of demersal fish and elasmobranchs" as a metric to populate indicator 1.6.1 "Condition of the typical species and communities". Indicator 1.6.1 is a "habitats" level indicator. Indicators derived from survey data of "mobile" species such as fish should be used to populate "species" or "ecosystem" level indicators. It may be more appropriate to use this metric to populate indicator 1.3.1 "Population demographic characteristics", a "species" level indicator. However, the MML as defined by CEC (2008) is a "community level" metric. Since it is not a species-specific indicator, it might be more appropriate to use this metric to populate the "ecosystem" level indicator 1.7.1.
18		Cross-application	Metrics that are applicable to more than one MSFD indicator are preferable - or metrics that are relevant to other directives (e.g. WFD).	Thus metrics of the abundance of different trophic guilds are relevant to indicators for both Descriptors 1 and 4. NB. When using the same metric under two or more Descriptors care should be taken in interpretation and thresholds. The same level may not indicate GES under different Descriptors

[1]. If prior information (e.g. Greenstreet *et al*., 2012; Bundy, *et al*., 2010) is not available, application of this particular criterion may require *post hoc* analysis to assess the level of indicator redundancy present within a suite of indicators, determine which indicators co-vary, and then to apply other criteria in the table to select the most useful indicator from among each co-varying group.

3.1.3 Process to evaluate metrics and indicators using criteria

WGECO realizes that the above criteria are very unlikely to be fully met by any one indicator. Each indicator is likely to have good and poor qualities. It is likely that an independent assessment of how well each indicator meets the criteria listed in Table 1 would be too burdensome to Member States (or others) so an evaluation against the Categories (Quality of underlying data, Responsiveness, Management, Conceptual and MSFD) above would seem more feasible. On a very simple basis, each indicator could be evaluated against these categories in terms of how well it meets the underlying criteria for that category. For example:

Table 2. Preliminary procedure to evaluate indicator criteria and score card.

Score	Evaluation outcome
Good	Indicator meets all or majority of underlying criteria (3 or 4 for those with 3 or 4, 4 or 5 for those with 5)
Moderate	Indicator meets at least two of underlying criteria
Poor	Indicator meets one or none of underlying criteria

This though is a very simplistic approach that may not be particularly helpful in decision-taking.

First, there is no standard categorization as to what extent an indicator might meet a criterion. For example, in Criterion 1 (*Metrics used to populate indicators should be easily and accurately measured or determined using technically feasible and quality assured methods. Quantitative measurements are preferred over qualitative, categorical measurements, which in turn are preferred over expert opinions and professional judgments*) it might be that accuracy for example may not be as great as desired; does this mean that the criteria is only partly met? It is probably necessary to describe this consistently (especially to ensure consistency between evaluations). WGECO is unaware of any consistent set of terms to describe these concepts, but notes that e.g. the International Panel on Climate Change does lay out standard terminologies in its area of work (see e.g. Mastrandea *et al*., 2010)

Second, it is likely that some of the criteria are more important than others and some may be “essential” while others would be “desirable”, and these may vary with each issue being considered. Thus it is important that the scientific soundness of all indicators is known (otherwise erroneous conclusions might be drawn from the indicator) but perhaps less important that the compatibility criterion is met, especially in some of the large and geographically heterogeneous subregions being used by MSFD. It is also possible that the importance of each indicator might vary geographically and between Member States. An example of such an evaluation is provided in Table 3.

Table 3. A preliminary evaluation of priority between criteria list in Table 1.

1	Metrics should be tangible	Desirable
2	Existing and ongoing data	Essential
3	Relevant spatial coverage	Desirable
4	Responds to change in ecosystem attribute	Essential
5	Responds to change in specific pressures	Desirable/information
6	Signal-to-noise ratio	Desirable
7	Relevant to management objectives	Desirable/information
8	Relevant to management measures	Desirable
9	Comprehensible	Essential
10	Established indicator	Desirable
11	Cost-effectiveness	Essential
12	Theoretically sound	Desirable
13	Early warning	Desirable
14	Indicator suites	Not relevant at this stage
15	Compatibility	Not relevant at this stage
16	State or pressure	Information
17	Metrics relevance to MSFD indicator	Essential
18	Cross-application	Desirable

3.1.4 Criteria to evaluate targets

The following section should be seen as even more of a work in progress than the previous one. The criteria described below, could more usefully be seen as principles for establishing criteria rather than actual criteria themselves.

According to the MSFD directive (Art. 3(7)), "an environmental target means a qualitative or quantitative statement on the desired condition of the different components of, and pressures and impacts on, marine waters." Therefore different types of targets are being established to capture collectively the state of ecosystem components (reflecting good environmental status of ecosystem components), impacts (reflecting the need to avoid or improve an undesirable state not equivalent to GES) and pressures (reflecting the need to reduce or stabilize them). Further, operational targets are those related to direct programme of measures to be established in order to reach GES. Each target might be associated with a threshold value between an acceptable and an unacceptable condition. The establishment of thresholds might be very critical and, in some cases speculative, unless formal procedures are available (e.g. fish stock related reference limits).

Where threshold values cannot be defined, the setting of trends-based targets can provide a pragmatic and operational alternative (Jennings and Dulvy, 2005; Shin *et al*., 2005). In essence this means that where scientific evidence suggests that current values of the indicator in question reflect a sub-GES situation, an *a priori* directional change can be proposed as an alternative to setting an absolute target indicator value. When such an approach is adopted, it is important to realize that meeting such trends-based targets does not mean that GES has been achieved. At best it implies that the appropriate measures have been put in place to move the ecosystem attribute reflected by variation in the indicator towards GES.

WGECO recognizes that general criteria based on relevant literature are available to assess state indicators (see sections above), but to our knowledge, no such evaluation procedures have yet been specified to evaluate the scientific soundness and feasibility of proposals for targets used to support the achievement of Good Environmental Status. The criteria WGECO propose might be applied to the indicators (Table 1) were therefore reviewed and revised to derive equivalent criteria that could instead be specifically applied to targets (additional criteria should also be considered, along the lines suggested above). While targets are inherently dependent on the indicators they are based upon, the characteristics that make them suitable for their use in the context of the MSFD are broader. In addition the OSPAR ICG-COBAM workshop report was examined and account taken of the proposed targets for the common indicators so far identified. Our criteria also took account of an analysis of the Commission Staff Working Paper on the Relationship between the initial assessment of marine waters and the criteria for good environmental status (SEC (2011) 1255 final).

Table 5. Very preliminary list of criteria by which to assess the suitability of targets being proposed by Member States for indicators stipulated for Descriptors 1 "Biodiversity", 4 "Foodwebs" and 6 "Seafloor integrity".

Number	Category	Characteristic	Criterion
1	Quality of underlying approach to set target	Methodological approach to define target should be consolidated	The approach used to define target should be based on recognized methodological standard and approaches.
2		Existing reference conditions	Reference conditions should be available to set thresholds. Quantitative thresholds are preferred over qualitative thresholds. Expert judgment thresholds should be avoided when possible and, when used, fully documented. The same logic applies to trends-based targets where absolute reference values are not known, but likely GES relative to current status is.
3		Relevant spatial domain	Targets should be defined in relation to large proportion of the assessment area. Target should point to reductions in the intensity and spatial extent of impacts on the marine environment in order to achieve or maintain GES.
4	Responsiveness-Representativeness	Environmental fluctuations and climate	Targets should be defined in order to take into account possible future changes due to natural variation also taking into account any evidence of climate change impacts.
5		Related to change in specific pressures	Targets should be associated to indicators on drivers (human activities) and related to specific pressures that have been clearly identified.

Number	Category	Characteristic	Criterion
6		Uncertainty	State-based targets, and related thresholds, should take into account uncertainty due to environmental fluctuations and climate change, as well as any uncertainty with estimating status.
7	Management	Relevant to management objectives	Target should be aligned with reductions in impacts that might be needed to achieve GES.
8		Relevant to management measures	Targets should be related to pressures that requires management. Pressure targets should provide a pragmatic focus on what is considered not to be a Good Environmental Status, and therefore facilitate the monitoring of progress towards achieving GES and the identification of appropriate programmes of measures
9		Comprehensible	Targets should be easily understandable by policy-makers and other non-scientists alike, and clear to communicate.
10		Established target	Targets already perceived by policy-makers and non-scientists as reliable and meaningful should be preferred over novel targets that perform the same role.
11		Pragmatic	Pressure targets should provide a pragmatic focus on what is considered not to be a Good Environmental Status. Pressure targets are mostly applicable to ecosystem components where data availability limits the capacity to derive "good" state indicators.
12	Conceptual	Theoretically sound	Scientific, peer-reviewed findings should underpin the assertion that reaching target will ensure GES to be achieved for a determinate ecosystem component.
13		Early warning of impending failure	When at GES, targets could have a threshold value such that deviation beyond this threshold away from GES conditions triggers an immediate management response.
14		Target suites	Target should be selected to avoid redundancy, increase the complementary on the GES components they rely on, and to ensure coverage of all key components.
15	MSFD specific	Compatibility	Targets should be capable of guiding progress towards achieving GES taking into account relevant existing targets laid down at national, EU or international level in respect to the same waters. They should be defined in the framework of regional cooperation
16		State, impact, pressure and operational targets	Targets should be clearly classified and distinguished between state, impact, pressure and operational targets and any confusion among them should be avoided

Number	Category	Characteristic	Criterion
17		Relevance to MSFD ecosystem components	Target should match to MSFD according to clearly defined ecosystem components in order to avoid confusion when assessing progress towards GES at the criterion level.
18		Cross-application	Where the same metrics are used to populate different MSFD indicators or targets for other policy drivers, the targets should ideally be the same in each situation.

The criteria in Table 5 are largely derived from those in Table 1. WGECO considered that it could also be useful to approach criteria for targets in an additional and independent way, perhaps using the analysis here to inform the approach. All targets can be considered to lie upon the axis of an indicator, and these indicators will have been evaluated against those criteria already (see above). So that any criteria here should ideally be *additional* to those for the indicators. Such additional properties might include:

- Achievability: how easy would it be to attain such a target;
- Being a known reflection of a publicly accepted GES or similar;
- Being compatible with other management obligations/targets e.g. in context of “favourable conservation status” under Habitats Directive);
- Being linked to known reference conditions.

WGECO would welcome inputs on other criteria which might be evaluated.

3.1.5 Targets evaluation process

A replication of the evaluation scheme suggested above for evaluating the performance of different indicators against the indicator criteria can be used as a preliminary method to assess the scientific soundness and feasibility of targets. However, the same concerns posed for indicator evaluation applies to the evaluation of targets, i.e. the lack of standard categorization as to what extent a target applies to criteria evaluation and the fact that some criteria are more important than others. Table 6 provided a first screening of the criteria suggested for target evaluation, giving a preliminary assessment of their relative importance.

Table 6. Preliminary list of priority for criteria by which to assess the suitability of targets being proposed by Member States.

Number	Characteristic	Priority
1	Methodological approach to define target should be consolidated	Essential
2	Existing reference conditions	Essential
3	Relevant spatial domain	Desirable
4	Environmental fluctuations and climate	Desirable
5	Related to change in specific pressures	Essential
6	Uncertainty	Desirable
7	Relevant to management objectives	Essential
8	Relevant to management measures	Essential
9	Comprehensible	Desirable
10	Established target	Desirable
11	Pragmatic	Desirable
12	Theoretically sound	Essential
13	Early warning	?
14	Target suites	Desirable
15	Compatibility	Desirable
16	State, impact, pressure and operational targets,	Essential
17	Relevance to MSFD ecosystem components	Essential
18	Cross-application	?

3.1.6 Relationships between indicator and targets

MSFD targets are related directly to the indicators. A first approach to combine the information provided by evaluation of both indicators and targets should take account this coherence. The soundness and feasibility of targets will be influenced by whether or not they are based on indicators that meet or fail the criteria for a good indicator. Therefore assessment of targets should first take account the evaluation of the indicators on which the targets are based. If indicator evaluation gives a moderate to poor score, then this would most likely infer that the related targets are not good, but the reverse does not necessarily hold true. Even targets based on a good indicator, could be evaluated as moderate to poor against the target criteria. This issue could be addressed by applying a matrix that combines the semi-quantitative scores assigned to both the indicator and target evaluations.

The lack of tangible information on indicators, targets and related thresholds impede the setting of a defined framework for set up at proper process at the time this document is being drafted.

3.1.7 Final comments

The discussions and preliminary results from WGECO support the preparation and positioning of ICES as a major player providing scientific support to the MS's MSFD implementation.

This is a several years' iterative process which should lead to useful methodologies and tools for the next implementation round (2018).

WGECO has attempted to lay down the basis for developing sets of criteria to evaluate the scientific soundness of feasibility of indicators and targets being proposed in support of the MSFD. Once the actual metrics being used by Member States to populate the various MSFD indicators have been established, these criteria can be applied to these specific proposed metrics and targets. It is important to note that at this stage, the first objective of such an evaluation would be to assess the performance of the two sets of criteria themselves, rather than to use the criteria to select, or otherwise, particular metrics and targets.

3.1.8 References

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3.2 Approaches for combining information provided by indicators and targets into an assessment of status

In most instances, assessment of status against GES targets at the regional seas scale will not be dependent upon a single indicator yielding a definitive "good" or "subgood" result. Here we consider two specific issues:

1) Deriving regional sea scale assessments of individual ecosystem component status for regions where no single systematic survey covers the whole region. Assessment of regional scale status will therefore have to rely on the information obtained from several disparate surveys, each covering different zones within the region, and each potentially using different survey methodologies and sampling gears.

2) Deriving overall assessments of regional sea scale status using numerous different indicators ("indicator suites") for particular Marine Strategy Framework Directive (MSFD) Descriptors, addressing status in several different ecosystem components.

The examples used here are all fish survey based, and relate to particular metrics of the status of fish communities. However, WGECO suggests that the questions that are addressed, and the outcomes that emerge, are relevant to many other MSFD indicators and ecosystem components.

3.2.1 Regional sea scale assessments from subregional sea scale surveys

The large fish indicator (LFI) supports the EcoQO for the "health" of demersal fish communities. The LFI is stipulated in the EC (2010) Decision document as an indicator to assess variation in the proportion of biomass occupying the top predator trophic level for

Descriptor 4 "foodwebs". The LFI was developed in the "Greater North Sea" (OSPAR region II) and is derived using the first quarter (Q1) international bottom-trawl survey (IBTS) (Greenstreet *et al*., 2011); a survey undertaken by several countries bordering the North Sea, but which is coordinated by ICES so that a single systematic methodology and sampling gear is used by all participants. Since this survey covers most of the "Greater North Sea" area, this single survey-based LFI can be used to derive a regional-sea scale assessment.

In other regional seas, for example OSPAR region III, the "Celtic Seas", the situation is different; the region is not covered by a single systematic survey, and regional sea-scale assessments will have to rely on the information provided from several different surveys, each covering a different zone within the region. In the "Celtic Seas" region three different surveys cover the majority of the area. The situation is eased by the fact that these surveys all use essentially the same type of trawl (but with different groundgears). However, quite different vessels are involved and the surveys are carried out at different times of year. Shephard *et al*. (2011) show how the methodology for determining the LFI developed in the North Sea (i.e. species suite selection and "large fish" length threshold selection, see also Section 7 (ToR e) can be applied to the Celtic Sea (the southern part of the "Celtic Seas" region) survey data to derive an LFI specifically "tuned" to assess the status of the particular demersal fish assemblage resident in the area. The same approach can be applied to each of the groundfish surveys operating in the "Celtic Seas" region to derive individual survey-based LFIs, each of which is "tuned" to monitor effectively any change in the demersal fish assemblage resident in the zone covered by each survey. This has been done, but trends in each of the survey-based LFIs are not consistent. The question then is how to "weight" the information derived from each individual survey to come up with a single overall, regional sea scale assessment.

Habitat variation across regional seas can give rise to distinctly different fish communities occupying different zones within the region, for example the roundfish-dominated fish assemblage in the deeper, muddy-bottomed northern North Sea and the flatfish-dominated assemblage in the shallower, gravel and sand bottomed southern North Sea (Fraser *et al*., 2008). Differences in the behaviour of individual survey-based indicators could be related to habitat differences between the zones covered by the different surveys. If so how do you weight the different indicators? Do you assume that each community is an entity in its own right and GES is required to be met in each of the individual survey-based indicators for GES at the regional sea scale to be achieved? Alternatively do you derive some composite indicator, for example an average of the individual survey-based indicators, and if so do you again assume that each community is an entity in its own right and equally weight each individual survey-based indicator? Or do you take account of differences in the area covered by each survey and weight the contributions of each individual survey-based indicator to calculation of the composite indicator on the basis of survey spatial range?

It is difficult to answer these questions based on an *a priori* rationale, good arguments can be made for each approach, but a more pragmatic empirical approach to obtaining an answer might also be found. Regional sea scale assessments of status (of fish communities) are possible in the "Greater North Sea" where the single systematic Q1 IBTS operates, and where there is good cause to expect that subregional scale variation in behaviour of different indicators, reflecting habitat variability, is likely (Fraser *et al*., 2008). In this case the regional scale survey data can be geographically partitioned to mimic different surveys. Deriving “tuned” LFI indicators for each geographic subdivision following the procedures established by Greenstreet *et al*. (2011) and Shephard *et al*. (2011), effectively replicating the process that would have to be applied in a region like the “Celtic Seas, the North Sea regional-scale LFI assessment can be “deconstructed” into several subdivisional, pseudo survey-derived, assessments. Various “reconstruction” procedures, (e.g. by simple averaging, biomass-weighted average, area-weighted average, etc.) can then be applied to explore which procedure best replicates the original regional-scale assessment.

There are other situations where different subregional scale surveys use different fishing trawls: otter trawls and beam trawls for example. This situation was explored in Section 7. The key message here is that even when a regional seas-scale survey is possible, this might actually not provide the most reliable assessment of regional scale status. Consider the situation where a regional-scale otter trawl derived LFI is primarily driven by changes in the roundfish dominated demersal fish community occupying the deep muddy-bottomed northern North Sea and variation in a regional-scale beam trawl derived LFI primarily reflects variation in the flatfish dominated demersal fish community occupying the shallow sand and gravel-bottomed southern North Sea. Only if both indicators show the same trend will use of one or other indicator provide a reliable assessment of regional scale status. If LFI trends obtained from the two surveys differ, then a composite assessment based on beam trawl data from the southern North Sea and otter trawl data from the northern North Sea might, ecologically, be a more appropriate basis on which to proceed.

3.2.2 Multiple indicators for single assessments

For Descriptors 1, 4 and 6, there are a total of 12 criteria and 23 indicators. The MSFD requires that GES be assessed for each criterion of each of the 11 Descriptors unless there are valid reasons not to. It is immediately clear that for some criteria, this assessment will be informed by two or more indicators. It is a widely held belief that suites of indicators should be used to monitor change in a variety of different attributes of ecosystem composition, structure and function. What is less clear is how variable trends in the different indicators that make up such suites should be interpreted to arrive at a single overall assessment of ecological status.

Recently a considerable amount of work has focused on deriving ecosystem assessments from suites of indicators (Bundy *et al*., 2010; Blanchard *et al*., 2010; Coll *et al*., 2010; Link *et al*., 2010; Shannon *et al*., 2010; Shin and Shannon, 2010; Shin *et al*., 2010a; Shin *et al*., 2010b). WGECO notes that this body of work exists and recommends that a review would be useful and could be reflected in next year’s terms of reference. However, even a preliminary examination of this literature suggests that a key aspect is the use of appropriate criteria to identify the most useful indicators with which to monitor variation in each attribute of ecosystem structure and function. In applying these criteria to select indicators, it was clear that only a single indicator was needed to represent each ecosystem attribute, so as to avoid indicator redundancy.

Indicator redundancy occurs when groups of metrics or indicators co-vary significantly, providing duplicate copies of a single signal rather than reflecting different independent signals (Greenstreet *et al*., 2012; Lyashevska and Farnsworth, 2012). For example, analysis of 15 indicators of demersal fish status in the North Sea indicated considerable covariation between them, and suggested that in fact only three structural and three compositional attributes were being monitored (Greenstreet *et al.*, 2012). When using a suite of indicators to assess ecosystem status, the issue of indicator redundancy becomes important when acceptable status (e.g. GES) is considered achieved if a stipulated proportion of the indicators meet their targets. Consider a suite of five indicators, and where acceptable status is considered to have been achieved if 80% of the indicators meet their individual targets. If in reality four indicators all co-vary and reflect change in one ecosystem attribute, all four are likely to meet their target more or less simultaneously and status would be assessed as acceptable. It would not be necessary for the fifth indicator to meet its target. But if in fact this fifth indicator reflected variation in a different ecosystem attribute, then acceptable status would be deemed to have been achieved when in reality only one of the two attributes monitored was in an acceptable condition. How to use potentially redundant indicators and how to base assessments that utilize potentially redundant metrics still raises questions that WGECO would wish to return to.

The whole issue is further compounded by the fact that in many instances, a particular MSFD indicator will be populated using numerous individual species-based metrics. For example current proposals for populating the population abundance/biomass indicator (1.2.1) in respect of demersal fish in the North Sea involves the assessment of biomass metrics for 27 "sensitive" fish species derived from Q1 IBTS data. This situation is, however, more straightforward. All that is necessary is an *a priori* "decision process" to determine how many species-specific metrics are required to meet their metric-level targets to consider that the indicator-level target has been achieved. WGBIODIV (ICES 2012) reviewed such a process applied to demersal fish survey-derived abundance data. This involved setting metric-level targets such that the probability of all species-specific metrics meeting their targets was the same, and known. Then the binomial distribution was used to identify the number of species-specific metrics needing to meet their metric-level targets for this to be a statistically significant departure from the expected number predicted by the binomial distribution.

3.2.3 References

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4 ToRb) DCF indicators

Examine and report on the DCF indicators of fishing pressures/impacts and possible developments/ improvements in these in the context of expected revision of the DCF and the operation of the MSFD i.e. (i) update the indicators in terms of technical details of how to do it and then make the actual analysis/calculation and (ii) provide possible input to ICES for the DCF revision.

4.1 Introduction

CEC 2008 provides a list of indicators to measure the effects of fisheries on the marine ecosystem (Table 4.1a). In the following section each indicator will be considered with a specific emphasis on the technical details required to calculate them as well as examples of the actual calculation.

Table 4.1a. DCF indicators (CEC, 2008).

Code specification	Indicator
1	Conservation status of fish species
2	Proportion of large fish
3	Mean maximum length of fishes
4	Size at maturation of exploited fish species
5	Distribution of fishing activities
6	Aggregation of fishing activities
7	Areas not impacted by mobile bottom gears
8	Discarding rates of commercially exploited species
9	Fuel efficiency of fish capture

4.2 Conservation status of fish species

This is an indicator of biodiversity to be used for synthesizing, assessing and reporting trends in the biodiversity of vulnerable fish species. Its purpose as a state indicator is to assess the performance of the Common Fisheries Policy at minimizing the impact of fishing activities on the marine ecosystem.

Calculating the Conservation Status of Fish (CSFa and CSFb) indicators is a multi-step process and users have had difficulty reproducing indicator series even when using what appear to be the same data and the same protocol. The STECF working group on the Ecosystem Approach to Fisheries Management (EWG 11–13, 2012) recently concluded that 'interpretation of these two indicators appeared very difficult, if not impossible in practice, leading to confused and incoherent signals; more work seems to be required before such indicators could be used to assess the conservation status of fish species within each ecosystem and efficiently contribute to set up a diagnosis on ecosystems health'. To address this problem, WGECO recalculated the CSF indicators for a case study area having strong data support (North Sea; 1983–2005). WGECO tried to reproduce a previously calculated time-series (MEFEPO, 2011) and commented on observed differences in protocol and results. The objective was to identify aspects of the process where potential for in-consistency arises. In addition, general comments on the interpretation and utility of the indicators are provided.

4.2.1 Method of calculation

The indicator is calculated in five steps according to the DCF regulation:

1) Species selection: list of species sensitive to fishing, i.e. large. Those species identified reliably of which >20 individuals are caught per year (all area), and of which L0.95 (the ninety-fifth percentile of the population length distribution) ≥40 cm are listed, of this list; the 20 largest species are the sensitive species.

2) Calculate the abundance index of individuals with length ≥L0.95/2 (a proxy for size-at-maturity).

3) On a ten years gliding window, calculate a decline index: the slope of a linear model; if the species is not rebuilt since (≥average abundance first three years): score the decline index according to the IUCN A1 criterion as follows:

• Min(decline) ≤90% 'critically endangered'	CR	3
• Min(decline) ≤70% 'endangered'	EN	2
• Min(decline) ≤50% 'vulnerable'	VU	1
• Otherwise 'least concern'	LC	0

The indicator is the average decline score across sensitive species; it varies from 0 (no species threatened) to 3 (all species critically endangered).

1) $y_{i,l}(t)$ catch of the population i by size class l, $t=t_1\dots t_f$
S total number of species

2) SV: Nsv sensitive populations = $\{L_{0.95,i} > 40\text{ cm} \;\&\; L_{0.95,i} > L_{0.95,j} \;\; \forall j \in (\text{S–SV})\}$;
Nsv = max(20, number of populations >40 cm)

3) $$a_i(t) = \sum_{l > L_{0.95,i}/2} y_{i,l}(t)$$

4) $t_k = t_1 \dots (t_f - 10)$:

$$a_i(t)/a_i(t_k) = \beta_{1,k} t + \beta_{2,k}, \quad t = t_k \dots t_k + 10$$

$\text{Id}_i = \text{score}(\min_k(\beta_k))$ (*Cf* tableau)

$\text{R}_i = \text{I}\{\exists t > t_{kmin} + 10 \;\; a_i(t) > A_i\}$ avec

$$A_i = \sum_{t=1}^{3} a_i(t)/3 \quad \text{ou} \quad A_i = \sum_{j=1}^{5} \max(a_i(t))/5$$

5) $$I = \sum_{i=1}^{Nsv} (1 - R_i) Id_i \Big/ N_{SV}$$

4.2.2 Calculation

Each step in the indicator protocol as described in the EC decision document (EC, 2008) is reproduced (see boxes). Related issues and questions arising from the WGECO analysis are then discussed.

Data source and survey area

Data required: Bottom-trawl survey data for relevant marine region. This indicator should be calculated using species, length and abundance survey data that have been collected from the largest proportion of the marine region over the longest available time period. The indicator would be survey specific. The methods require that surveys are conducted annually in the same area with a standard gear.

The WGECO analysis focused on the North Sea quarter one (Q1) International Bottom-trawl Survey (IBTS). Data were downloaded from the ICES DATRAS website (www. http://datras.ices.dk/) and comprised 'Exchange' files. The CSFa indicator requires a continuous time-series, but at the time of downloading (April 2012) North Sea Q1 data for 2006 were not available on DATRAS. Hence, the CSF indicators were calculated for the period 1983–2005.

The spatial coverage of a survey may be modified over time as primary objectives and data needs change. However, spatial heterogeneity in the underlying fish community can drive variation in community indicators that may confound the temporal signal. Hence, a standard survey area (footprint) should be defined. Following the protocol of Greenstreet *et al*. (2011) and Shephard *et al*. (2011), WGECO excluded ICES statistical rectangles sampled in <50% of all years (Table 4.1). A related problem can arise when the survey makes substantial changes in methodology (e.g. measuring and subsampling) or hardware (e.g. vessel, gear). It is important to ensure that these issues are identified in any survey-series to be analysed.

Table 4.1. ICES statistical rectangles excluded due to inconsistent sampling (sampled in <50% of all survey years).

28F0	30E2	40G2	50E8
29F0	30F0	46E7	
29F1	30F1	46G1	
	31F1	47E7	
	31F3	49E8	
	35F5		
	36F8		

Standard species suite

Calculation of indicator: This is a two stage process where the species to include in the indicator are identified and then used to build a dataset for calculating indicator values.

When calculating the indicator, species should be excluded if:

- They have morphology, behaviour or habitat preferences that are expected to lead to low and variable catchability in the survey gear (this does not exclude species that should, in theory, be effectively sampled by the gear but which have become so scarce that they are now caught infrequently; unless excluded under '2' below).
- Mean annual catch rates of the species in the entire survey area over the entire survey period are less than 20 individuals (of any length).
- They have an asymptotic total length (L_∞) and/or maximum recorded total length of <40 cm.

- They cannot be identified reliably (although all practicable effort should be made to ensure species-level identification).

Greenstreet *et al*. (2011) and Shephard *et al*. (2011a) provide a clear protocol for defining the fish species suite for calculation of the Large Fish Indicator (LFI). This protocol includes consideration of the ecological role of each species and its catchability in the survey gear; species that are not representatively sampled are excluded. Following this precedent, WGECO excluded all 'non-fish' species (e.g. shellfish and squid), since these are not part of the fish community of interest. Several (mainly pelagic) fish species (Table 4.2) were also excluded based on poor (non-representative) sampling. Notably, the appropriate fish species suite differs among regions and surveys. The values for maximum recorded total length L_{max} used by WGECO were derived from the IBTS and are shown in Table 4.3.

Table 4.2. Fish species excluded on the basis of catchability in the survey gear.

Sprattus sprattus
Sardina pilchardus
Micromesistius poutassou
Clupea harengus
Scomber scombrus
Sander lucioperca
Engraulis encrasicolus
Trachurus trachurus
Salmo trutta
Salmo salar

Table 4.3. L_{max} values used to calculate CSF indicators.

Species	Lmax (cm)	Species	Lmax (cm)	Species	Lmax (cm)
Acipenser sturio	500	*Glyptocephalus cynoglossus*	73	*Platichthys flesus*	60
Agonus cataphractus	21	*Gymnammodytes semisquamatus*	37	*Pleuronectes platessa*	122
Alosa alosa	83	*Helicolenus dactylopterus*	47	*Pollachius pollachius*	130
Alosa fallax	73	*Hippoglossoides platessoides*	52	*Pollachius virens*	130
Amblyraja radiata	105	*Hippoglossus hippoglossus*	300	*Pomatoschistus lozanoi*	8
Ammodytes marinus	25	*Hyperoplus lanceolatus*	49	*Pomatoschistus microps*	9
Ammodytes tobianus	24	*Lampetra fluviatilis*	50	*Pomatoschistus minutus*	11
Anarhichas lupus	150	*Lepidorhombus whiffiagonis*	60	*Pomatoschistus pictus*	6
Anguilla anguilla	133	*Leucoraja naevus*	71	*Psetta maxima*	122
Argentina silus	85	*Limanda limanda*	49	*Raja brachyura*	125
Argentina sphyraena	43	*Lophius budegassa*	122	*Raja clavata*	139
Arnoglossus laterna	31	*Lophius piscatorius*	244	*Raja fyllae*	60
Belone belone	93	*Lumpenus lampretaeformis*	50	*Raja montagui*	80
Brosme brosme	120	*Lumpenus lumpretaeformis*	50	*Sardina pilchardus*	34
Buglossidium luteum	15	*Melanogrammus aeglefinus*	112	*Scomber scombrus*	67
Callionymus lyra	30	*Merlangius merlangus*	70	*Scophthalmus maximus*	122
Callionymus maculatus	16	*Merluccius*	140	*Scophthalmus rhombus*	75
Callionymus reticulatus	11	*Merluccius merluccius*	140	*Scyliorhinus canicula*	100
Ciliata mustela	25	*Microchirus variegatus*	43	*Scyliorhinus stellaris*	170
Clupea harengus	55	*Micromesistius poutassou*	50	*Sebastes marinus*	100
Clupea harengus	55	*Microstomus kitt*	65	*Sebastes viviparus*	35
Conger conger	300	*Molva molva*	200	*Solea solea*	85
Cyclopterus lumpus	43	*Mullus barbatus*	37	*Solea vulgaris*	85
Dicentrarchus labrax	103	*Mullus surmuletus*	49	*Sprattus sprattus*	20
Dipturus batis	285	*Mustelus asterias*	140	*Squalus acanthias*	160
Echiichthys vipera	18	*Mustelus mustelus*	200	*Taurulus bubalis*	25
Enchelyopus cimbrius	41	*Myoxocephalus scorpius*	60	*Trachinus draco*	53
Engraulis encrasicolus	24	*Myxine glutinosa*	80	*Triglops murrayi*	24
Eutrigla gurnardus	60	*Osmerus eperlanus*	45	*Trisopterus esmarkii*	35
Gadiculus argenteus	15	*Pholis gunnellus*	31	*Trisopterus luscus*	46
Gadus morhua	200	*Phrynorhombus norvegicus*	15	*Trisopterus minutus*	40
Gaidropsarus vulgaris	60	*Phycis blennoides*	110	*Zeugopterus punctatus*	31
Galeorhinus galeus	195			*Zeus faber*	90
				Zoarces viviparus	52

Having formatted the dataset, a useful check is to try to reproduce a published indicator series. WGECO used the formatted data to calculate the LFI for the North Sea Q1 survey and found the result to be close to that of Fung *et al*. (2012).

Reference species list

The following process should be used to select species and size classes when calculating the indicator:

1) Compile a list of species recorded in the history of the survey and their mean asymptotic total length (L_∞) and/or maximum recorded total length (if ≥40 cm). Asymptotic total length or maximum recorded total lengths are ideally determined from total length and age data collected on the same survey. A mean value for the survey period should be used when there are multiple estimates of L_∞, but the highest recorded value of maximum total length should be used.
2) Rank the species listed under '1' from high to low asymptotic total length (L_∞) and/or maximum recorded total length (use maximum total length only in those cases when L_∞ cannot be calculated from available size at age data).
3) Select the 20 largest species by total length (or all the species in the list if <20) from the rankings produced in '2'. Once this list has been defined it should be used for calculating indicator values in all subsequent years.

Following MEFEPO (2011), WGECO defined the list of reference fish species based on the first five years of survey data. For annual abundance, WGECO used observed numbers-at-length for each species in the 'Exchange' data. This approach avoids bias associated with raising catches by haul duration or swept-area. Notably, because all 20 species may not appear in all survey years, a longer list of 'candidate' species can be useful at this stage. Similarly, in some surveys there may not be 20 species for which L_{max} is >40 cm and some other length threshold or number of species would be required. The final list is defined as those species that occur in an 'adequate number' of survey years. The protocol allows for filling in of some missing years using the minimum observed abundance/2. However, at some point, this correction is likely to bias the indicator and so an objective threshold presence is demanded. As an arbitrary first-order threshold, WGECO retained species that were present in >50% of all analysed survey years.

Mean annual species abundance

1) For each of the species identified in '3' calculate mean catch rates, standardized to account for any changes/differences in tow duration (e.g. number per hour) for individuals of length ≥0.5 L_∞ only.

Despite a standard IBTS protocol, there is often considerable variation among individual survey tows and some standardization of the data is typically required. MEFEPO (2011) used the DATRAS 'Cpue_Numbers_Per_Hour' index in which catch by species and length group in each survey tow is simply raised from observed tow duration to 60 min. This index does not account for differences among survey tows in the area of seabed swept by the fishing gear; swept-area varies with depth, weather conditions and other factors. WGECO used unprocessed 'Exchange' data and calculated the indicators using an alternative standardization approach, i.e. number of fish by species and length class per km^2 swept-area. This approach accounts for variation in swept-area among tows but is still susceptible to bias associated with raising observed catches.

Of the 22 fish species in the candidate list, only 18 species occurred in >50% of survey years and were retained as the final reference species list. This list was compared (Table 4.4) with the species list of MEFEPO (2011). There are considerable differences, which are likely a consequence of differences in the underlying survey data and possibly from a different threshold at which species were deemed to have too many years of zero abundance. Differences in L_{max} values applied to each species would also affect definition of the species list.

Table 4.4. Fish species used by MEFEPO and WGECO to calculate CSF indicators for the North Sea Q1 survey. Species common to both analyses are highlighted in bold.

Species	WGECO	MEFEPO
Amblyraja radiata	**x**	**x**
Anarhichas lupus	**x**	**x**
Brosme brosme	x	
Cyclopterus lumpus		x
Enchelyopus cimbrius		x
Eutrigla gurnardus		x
Gadus morhua	**x**	**x**
Glyptocephalus cynoglossus	**x**	**x**
Hippoglossoides platessoides		x
Leucoraja naevus	**x**	**x**
Lophius piscatorius		
Melanogrammus aeglefinus	**x**	**x**
Merlangius merlangus	**x**	**x**
Merluccius merluccius	x	
Microstomus kitt		x
Molva molva	x	
Pleuronectes platessa		x
Pollachius pollachius	x	
Pollachius virens	**x**	**x**
Psetta maxima	x	
Raja clavata	x	
Raja montagui	x	
Scophthalmus rhombus	x	
Scyliorhinus canicula	x	
Solea solea		x
Squalus acanthias	**x**	**x**

Calculating the CSF indicators

Two indicators of the biodiversity of vulnerable fish species can be calculated from data compiled according to the preceding process: (a) an indicator of the biodiversity of vulnerable fish species that responds to changes in the proportion of contributing species that are threatened and (b) an indicator of the biodiversity of vulnerable fish species that tracks year-to-year changes in the abundance of contributing species. Both indicators assume that the survey catch rate provides an index of abundance.

h) For each species, catch rates in the first year of the survey are compared with catch rates ten years later. To achieve this a linear model is fitted to the first x years of data, $t_1 - t_x$ and to each successive year, i.e. $t_1 - t_{x+1}$, $t_1 - t_{x+2}$,…, $t_1 - t_{maximum}$, where $t_{maximum}$ is the final year for which data are available. The percent change in catch rate of the species is then calculated from the initial (t_1) and final (t_x to $t_{maximum}$) catch rate as predicted from the least-squares linear model fit. Species that meet any one of the decline criteria in any year of the time-series are categorized as threatened; unless their numerical catch rate subsequently increases above a preset catch rate threshold. This should be taken as the mean catch rate over the first three years of the time-series. The composite threat indicator is then calculated for each year as the average of the species threat scores (critically endangered if ≥90% decline- score =3, endangered if ≥70% decline- score=2, vulnerable if ≥50% decline- score=1) and allocated to the final year of the period over which the decline was measured. The indicator value is readily interpreted because the scores can vary from 0 to 3, such that a score of 0 is equivalent to no species meeting any of the threat criteria and a score of 3 is equivalent to each species being critically endangered.

Following Dulvy *et al*. (2006), WGECO used a 10 yr 'moving window' (results are moderately sensitive to the width of the time window used in the calculation). Specifically, the linear model was fit to the first 10 years of data, $t_1 - t_{1+x}$, and to each successive year, i.e. $t_2 - t_{2+x}$, $t_3 - t_{3+x}$,…, $t_{maximum-x} - t_{maximum}$. This approach reduces the influence of decisions about time-series length since species abundance trends are not driven by a specific fixed starting point. However, the problem of starting point remains because of the use of a preset catch rate threshold, taken as the mean catch rate over the first three years of the time-series. If the series started in a time period when species abundances were very low, then any positive change will suggest recovery even for species that are still objectively 'critically endangered' according to IUCN criteria (although IUCN recently released new criteria for exploited species). An alternative threshold is the average of the *n* highest values in the time-series; although this doesn't overcome the problem for strongly depleted species. A time-series of the CSFa indicator was calculated (Figure 4.1a) and compared and with that of MEFEPO (2011) (Figure 4.1b).

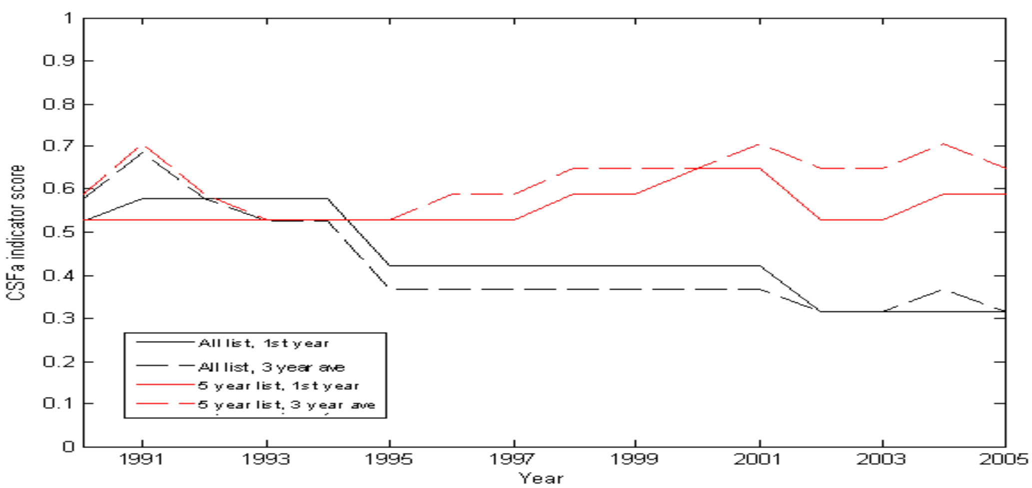


Figure 4.1a. CSFa indicator for the North Sea IBTS calculated with and without the species list being fixed to the first five years data, and with the reference period as either the first or average of the first three years data (MEFEPO, 2011). An increase in the indicator value indicates a decline in conservation status.

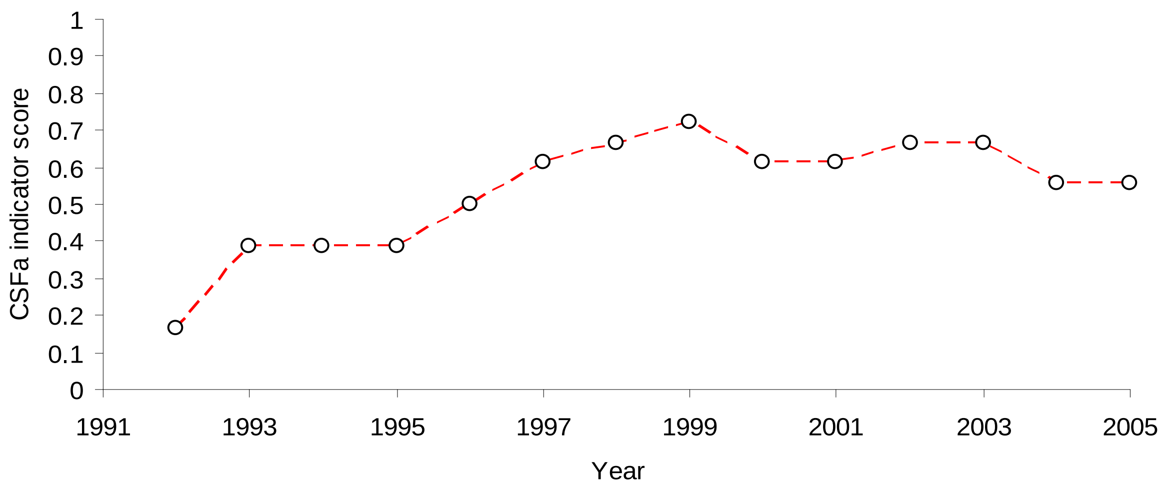


Figure 4.1b. CSFa indicator for the North Sea IBTS calculated with the species list being fixed to the first five years data, and with the reference period as the average of the first three years data. This is the series calculated by WGECO and corresponds to the dashed red line in Figure 4.1a.

i) Catch rates in a given year are expressed as a proportion of the mean catch rate in the first three years of any given survey (for which the mean catch rate is defined as 1). In any given year, the indicator is calculated as the geometric mean of relative adult numerical abundance. When calculating the geometric mean, proportions are log transformed as log(x+a), where x is the proportion and a is 0.5 times the minimum non-zero proportion in the time-series.

WGECO found this text hard to follow. More simply, this part of the protocol demands:

1) Calculate annual abundance (standardized by haul duration or swept-area) for each species;

2) Logtransform both annual species abundance and reference species abundance (log(x+a) where x is the relative abundance and a = 0.5 times the minimum non-zero species abundance in the time-series;

3) Express logtransformed annual species abundance as a proportion of logtransformed reference species abundance;

4) The indicator is then calculated as the geometric mean of these species proportions for each year.

A time-series of the CSFb indicator was calculated (Figure 4.2a) and visually compared with that of MEFEPO (2011) (Figure 4.2b). The CSFb indicator is simpler to calculate than CSFa but includes a further step in which the outcome may be changed as a function of previous decisions about the dataset. Values for mean annual fish species abundance are logtransformed using the minimum observed value a in the time-series. This transformation creates a similar problem to that noted above concerning replacing zero values for annual abundance of given species using the minimum observed abundance/2. Both corrections use values which are derived from the data and hence susceptible to definition of standard survey area and starting time period.

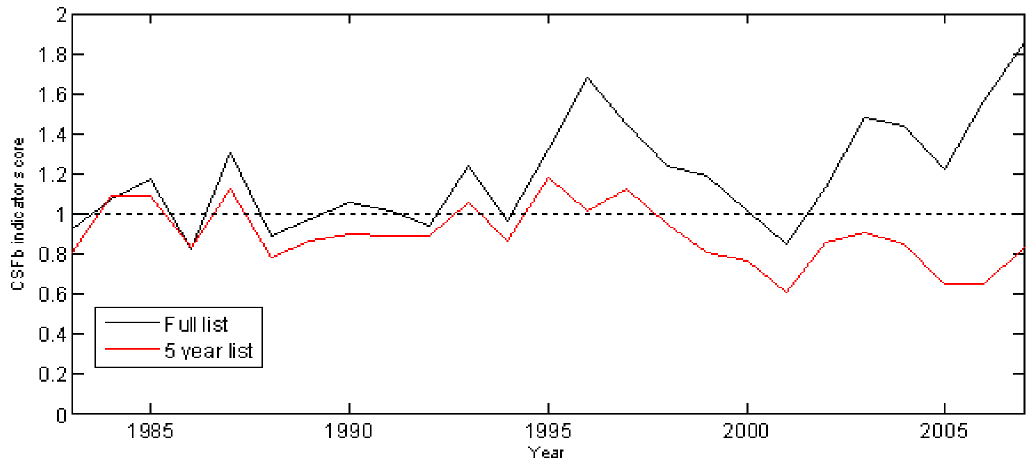


Figure 4.2a. CSFb indicator for the North Sea IBTS calculated with and without the species list being fixed to the first five years data (MEFEPO, 2011). A decrease in the indicator value indicates a decline in conservation status.

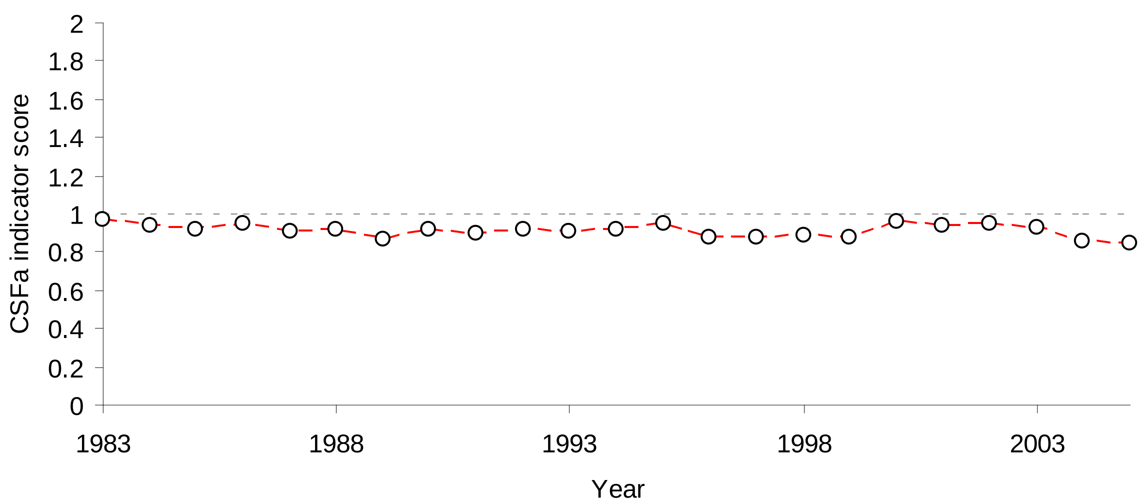


Figure 4.2b. CSFb indicator for the North Sea IBTS calculated with the species list being fixed to the first five years data. This is the series calculated by WGECO and corresponds to the solid red line in Figure 4.2a.

Comparison of time-series

The general trends in both indicators time-series appeared similar between MEFEPO (2011) and WGECO although absolute values differed. A summary of possible sources of divergence is given in Table 4.5, many of these issues are common to calculation of any indicator from fisheries survey data but some are specific to the CSF indicators.

Table 4.5. Steps in the CSF indicator process which can affect the outcome.

Calculation step	'Issue'
Common to use of survey data	
Defining survey period	Changes in survey protocol may not be obvious but can bias species abundances
Defining survey area	Survey coverage can vary in relation to a spatially heterogeneous fish community
Defining fish species suite	Decisions on survey catchability and ecological importance may be qualitative and inconsistent
Calculating mean annual abundance by hour or swept-area	Both methods of 'raising' the survey data create potential bias
Referring to the CSF indicator	
Applying L_{max} to each species	L_{max} values vary with region and survey and 'local' may be unavailable in some cases
Excluding species based on minimum abundance (>20 individuals yr-1)	Cpue indices may bias annual abundance estimates for uncommon species and length groups
Replacing zero values for annual abundance with min observed value/2	Min abundances may be biased by cpue indices and/or survey spatial/temporal definition
Excluding species for which there are 'too many' zero values	Exclusion or inclusion of certain species can change the indicator
Logtransformation of abundances using min annual abundance a/2	Min abundances may be biased by cpue indices and/or survey spatial/temporal definition

4.2.3 Synthesis and recommendations

Problems with meaning and interpretation of the CSF indicators

The proposed reference direction for indicator (a) is a significant reduction in the rate of increase, consistent with the WSSD target of achieving a significant reduction in the rate of biodiversity loss (by 2010). A decrease in the value of the indicator would also show progress towards the Common Fisheries Policy objective of ensuring that the impacts of fishing on marine ecosystem are sustainable. A limit reference point for this indicator would be 1 (when all species are listed as 'vulnerable' on average).

Benefits of indicator (a): Values of the indicator can be linked directly to the IUCN process for identifying critically endangered, endangered and vulnerable species. The indicator is therefore consistent with other threat-based indicators used to report on the status of mammals, birds and amphibians and which are used to track progress in relation to the WSSD biodiversity commitments. ICES assessed stocks that meet these simple but widely used threat criteria have been shown, without exception, to be exploited beyond safe biological limits (note that the decline associated with 'vulnerable' exceeds that which would be required to achieve MSY and that the declines associated with 'endangered' and 'critically endangered' would place stocks at risk of reduced reproductive capacity). It is also possible to set limit reference points and reference directions for this indicator.

The proposed reference direction for indicator (b) is a significant reduction in the rate of decline, which would be consistent with the WSSD target of achieving a significant reduction in the rate of biodiversity loss (by 2010). An increase in the value of the indicator would show progress towards the Common Fishery Policy objective of ensuring that the impacts of fishing on marine ecosystem are sustainable.

Benefits of indicator (b): Values of the indicator track inter annual changes in the catch rates of the larger, and therefore more vulnerable, species in a fish community. Reference directions can be set for this indicator.

The relative abundance of fish species in a marine community fluctuates subject to various intrinsic and extrinsic factors; variation in species evenness. In this dynamic context, the impact of fishing on biodiversity depends on the response of sensitive species. Typically, species having the most vulnerable life-history traits will decline rapidly (Jennings *et al*., 1998; Gislason *et al*., 2008; Le Quesne and Jennings, 2012) and can become functionally extinct (Dulvy *et al*., 2003); a reduction in species richness. The most useful (responsive and specific; Rice and Rochet, 2005) indicators of fishing pressure will thus respond to the species loss signal and minimize noise from 'natural' environmental and community-level dynamics.

The CSFa indicator integrates temporal trends in relative abundance of the suite of fish species in a given survey dataset that have greatest L_{max}. This size component of the community typically includes vulnerable species such as elasmobranchs and slow-growing teleosts. The indicator thus comprises a univariate metric capturing some aspects of the status of vulnerable fish species. However, WGECO question whether the CSFa actually expresses the predicted fishing impact on the community. The CSFa indicator averages vulnerability scores that are contingent on fitted abundance derived from linear models fitted through a moving window of ten data points for each species. By averaging abundance trends for (approximately) 20 species the indicator certainly incorporates decline and extinction, but the complete loss of a few species could theoretically be hidden by natural fluctuations in the other species. This may represent failure in 'specificity', i.e. a poor signal to noise ratio. In this context, the effect on the indicator of the addition of further individual annual data points will be minimal unless there is some marked change in abundance, which could simply reflect noise. Recalculating the indicator in longer time increments may be more revealing.

The method also presents some statistical problems. Least-squares models are fitted to successive ten year series of survey abundance data. Such short dataseries are highly susceptible to both environmental stochasticity and to survey sampling error such that fish abundance trends are unlikely to be monotonic (linear). Where trends are not linear, the least-squares model may be inappropriate and the slope hard to interpret. Conversely, an advantage of the short time-series is that highly complex trajectories are less likely. Since results are moderately sensitive to the width of the time window used in the calculation, definition of an optimum window width may be demanded for given surveys.

Even assuming that the linear model is valid, all species abundance slopes are taken into account in the indicator calculation, regardless of their statistical significance. Given variability of survey abundance indices and short time windows, a slope might be high but not statistically significant, and thus not meaningful. An objective criterion, i.e. significance threshold, would be valuable here to determine which slopes should be incorporated into the calculation. A summary of those aspects of the CSF indicators demanding consideration is given in Table 4.6.

Recommendations

The experience of WGECO raises questions about the applicability of the CSF indicator to the MSFD process. Essentially, there is a lot of scope for error (or at least interpretation) in implementing the protocol. There are also difficulties with averaging vulnerability scores from short-term (10 yr) abundance trends of 20 species. Assuming that species trends are robust, it remains unclear as to what an 'average' state among vulnerable fish species may mean. An alternative approach would be to retain abundance information at the individual species level and employ some objective statistical protocol for deriving an indicator from these discrete series. There may be scope for a method that could objectively determine how many of the fish species in a 'vulnerable' species suite should be showing positive abundance trends in order for an overall positive trend to be assumed. An example of such an approach is given in ICES 2012 WGBIODIV**.** In the short term, a simpler and more easily interpretable way of describing common trends among sensitive species would be valuable.

Despite the potential problems identified above, WGECO note that both the CSF time-series for the North Sea seem to a slow decline in fish conservation status; consistent with the expectation that there is a decline in fish 'biodiversity' in this system (Dulvy *et al.*, 2006). However, the CSFa indicator remained below the reference value of 1 for the whole period, suggesting that despite the decline that state is still considered 'acceptable' from an IUCN perspective. Given the complexity of the calculation protocol and challenges in interpretation, WGECO question the suitability of the CSF indicators in monitoring of marine fish communities under the MSFD. Alternative approaches are under development (see ToR A in this document and ICES 2012 WGBIODIV) should be compared to the CSF indicator before making a final decision on the most appropriate indicator for biodiversity as part of the DCF.

Table 4.6. Aspects of the CSF indicators that require consideration.

'Issue'	Possible solution
Linear model may be invalid	?
Slopes may not be statistically significant	Some criterion for acceptance, e.g. significance threshold
Results are highly dependent on the length of the time-series (especially the starting point).	?
Results are moderately sensitive to the width of the time window used in the calculation	Window width can be defined depending on how conservative one wants to be
It may not be appropriate to recalculate the CSFa indicator using single year increments.	Use multiyear increments?
Species abundance may be zero in many years	Some criterion for species retention, e.g. percentage of years present
In ecosystems other than the North Sea there might not be 20 species having L_{max} >40 cm and meeting all the criteria.	Use a shorter list of species? Or change the length threshold?

4.3 Proportion of large fish

Refer to existing work/literature.

4.3.1 Method of calculation

The calculation process is reproduced (see text below) as described in EC (2008).

The indicator can be calculated for the entire assemblage that is caught by that particular gear or a subset based on morphology, behaviour or habitat preferences (e.g. bottom-dwelling species only).

The "large" fish threshold needs to be set at a level that decreases the noise around the trend caused by e.g. recruitment effects while maintaining the indicators' sensitivity. In the IBTS North Sea dataset a threshold of 40 cm was used, which amounted to between 5800 and 25 000 fish being sampled in each year. Across the whole time-series fish over 40 cm represented over 0.5% of the total number of fish sampled.

The proportion of "large fish" is calculated as: $P_{>40cm} = \frac{W_{>40cm}}{W_{Total}}$ where $W_{>40\,cm}$ is the weight of fish greater than 40 cm in length and W_{Total} is the total weight of all fish in the sample.

4.3.2 Calculation

WGECO calculated or collated (STECF, 2012) time-series of the 'Proportion of large fish' (Large Fish Indicator; LFI) for several marine regions (Figure 4.3.2.1). The standard protocol from EC (2008) (text above) was used.

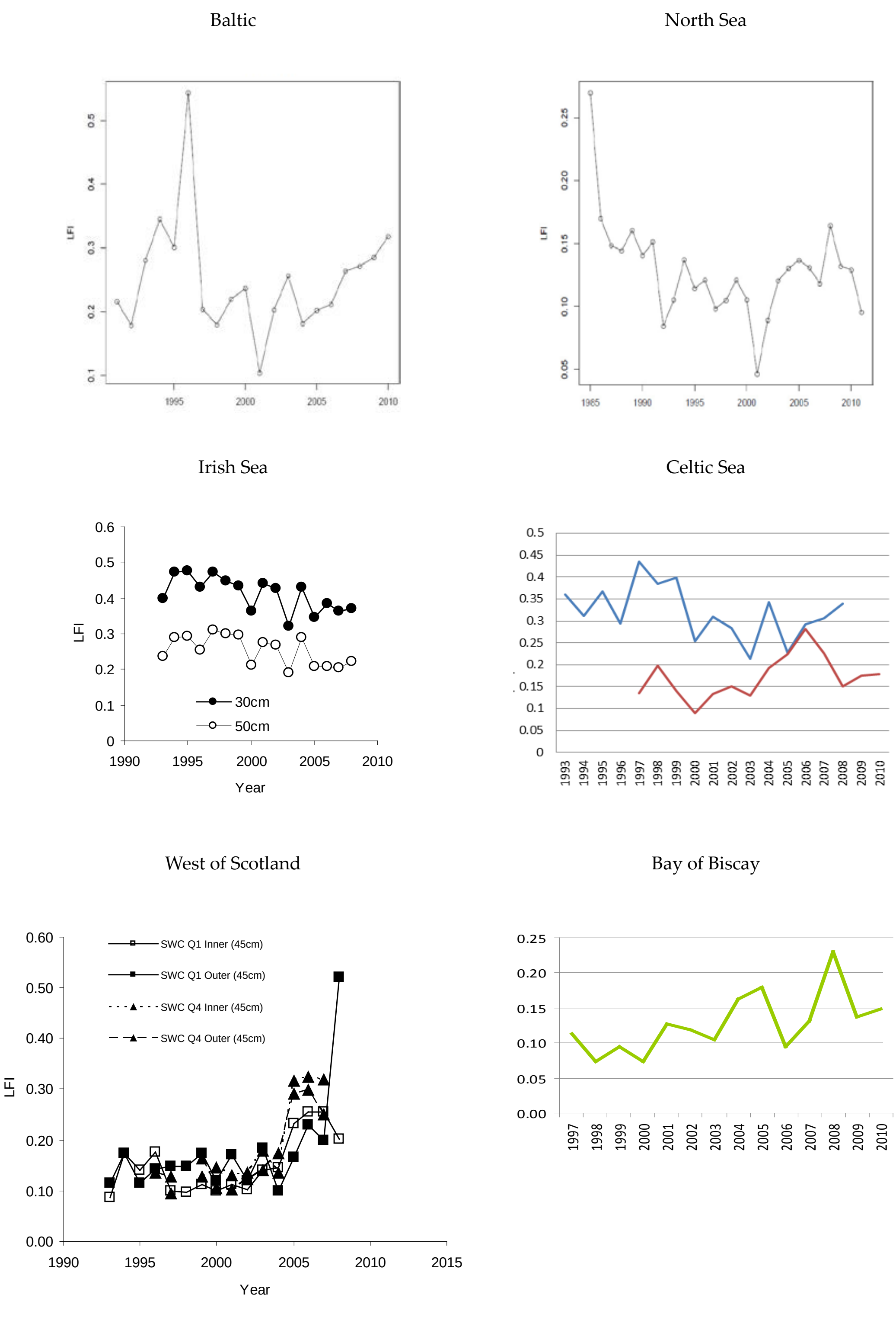


Figure 4.3.2.1. Time-series of the 'Proportion of large fish' indicator for various marine regions.

4.3.3 Synthesis and recommendations

Much work has gone into the development of this indicator and it is now widely applied in various regions. We identified no new major issues that need to be considered when calculating this indicator.

When calculating the indicator for other regions or based on different surveys it should be realized the threshold applied (in the original methodology 40 cm for the North Sea IBTS) is both region- and survey-specific and should be first determined before the indicator can be calculated.

4.4 Mean maximum length of fishes

4.4.1 Method of calculation

The calculation process is according to EC (2008).

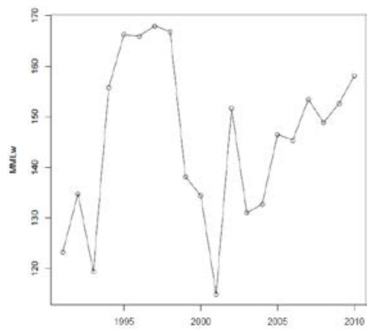
The indicator can be calculated for the entire assemblage that is caught by a particular gear or a subset based on morphology, behaviour or habitat preferences (e.g. bottom-dwelling species only).

Mean maximum length is calculated as: $\overline{L_{\max}} = \sum_j (L_{\max j} N_j) \Big/ N$ where $L_{\max j}$ is the maximum length obtained by species j, N_j is the number of individuals of species j and N is the total number of individuals. Asymptotic total length (L_∞) is preferred to maximum recorded total length if an estimate is available, but it is recognized that such data may not be available for many species.

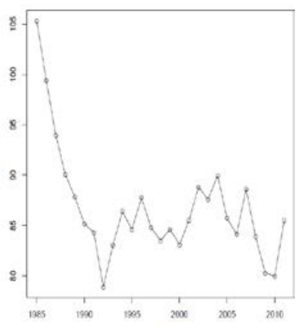
4.4.2 Calculation

WGECO calculated or collated (STECF, 2012) time-series of the 'Mean maximum length (by biomass) of fishes' for several marine regions (Figure 4.4.2.1). The standard protocol from EC (2008) (text above) was used.

Baltic



North Sea



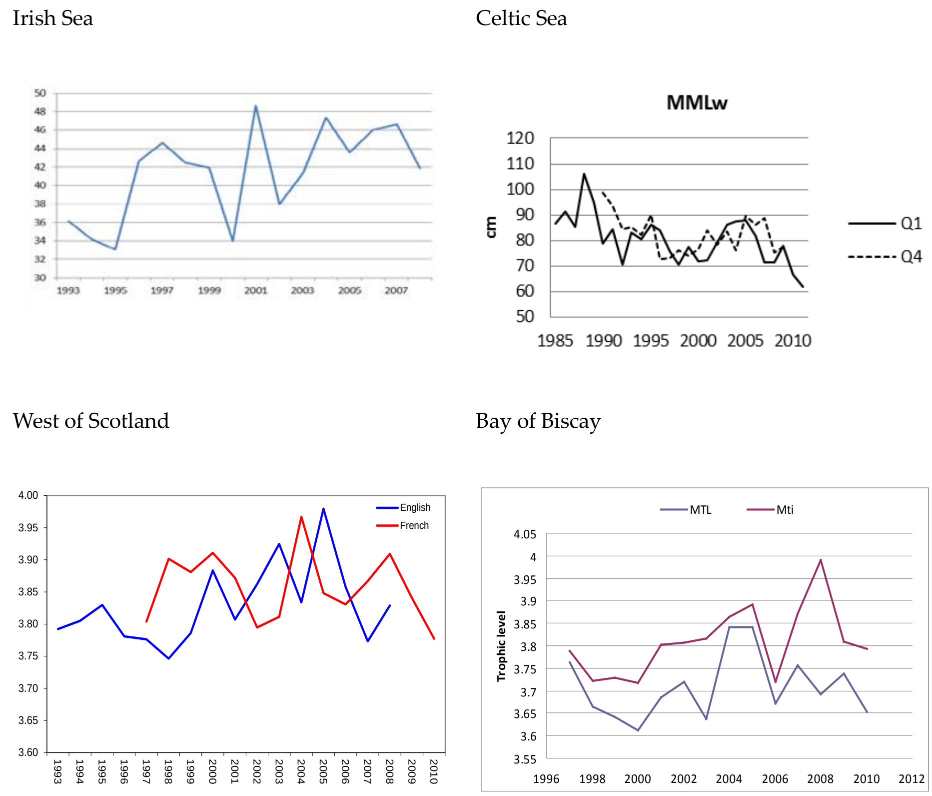


Figure 4.4.2.1. Time-series of the 'Mean maximum length (by biomass) of fishes' indicator for various marine regions.

4.4.3 Synthesis and recommendations

Application of the methodology to calculate the indicator in the different regions did not reveal any new major issues that need to be considered when calculating this indicator.

When using this indicator it should be recognized that this indicator is intended to reflect changes in the species composition driven by differences in life-history characteristics, NOT the size structure.

4.5 Size at maturation of exploited fish species

This indicator is meant to track the potential "genetic effects" of fishing on exploited populations. It should contribute to assessing the performance of CFP in relation to the objectives of 'minimizing the impact of fishing activities on the marine ecosystem'.

4.5.1 Method of calculation

The indicator is the probabilistic maturation reaction norm (i.e. the probability of maturing) and this is derived from the maturity ogive (i.e. the probability of being mature) and from the mean annual growth at age as:

$$m(a,s) = (o(a,s) - o(a-1, s-\Delta s(a)))/(1 - o(a-1, s-\Delta s(a)))$$

where a is age, s is length, $o(a,s)$ is the maturity ogive, and $\Delta s(a)$ is the length gained from age $a-1$ to a. Estimation of the probabilistic maturation reaction norm thus requires (i) estimation of maturity ogives, (ii) estimation of growth rates (from length-at-age), (iii) estimation of the probabilities of maturing, and (iv) estimation of confidence intervals around the obtained maturation probabilities.

These calculations have to be carried out based on fisheries-independent data with measurements of age, length, sex and maturity status (immature or mature) for the same individual. To obtain reliable estimates, individual measurements should be taken for at least 100 individuals per age class per year in the population. These age classes need to contain both juvenile and adult individuals. The necessary sample size in a given year can thus be derived from the number of age groups in the population.

The DCF specifications further stress that owing to the sampling requirements the indicator is best applied to species that are already subject to stock assessment. If resting individuals (i.e. individuals that are mature but do not spawn in the sampled season) can be mistaken for immature individuals, they need to be classified as juvenile, an adult that spawn(ed) within the season or a resting adult individual. A requirement of the monitoring programme is that it covers a large enough part of the marine region and/or the population for which the indicator is calculated. Data collected during the existing DCR surveys should be used to calculate this indicator.

4.5.2 Issues

The indicator specifications outline the substantial amount of data required for the calculation of this indicator for one single species. This implies the indicator can only be calculated for a fairly limited number of species; it may not even feasible for all species subject to stock assessment to measure, age, and determine sex and maturity for a sufficient number of individuals each year, owing to the cost of these data, and the fact that many species might not be maturing at the time of the DCR surveys. Experience with routine acquisition of maturity data for a large number of species in MEDITS (Mediterranean groundfish trawl survey) suggests that in the end, reliable maturity data can be acquired for a very limited number of species in each area. Therefore it is not likely that this indicator will inform about “the impact of fishing on the marine ecosystem;” rather, on a restricted number of target species.

However, the most important issue with this indicator is the interpretation of trends. The maturation reaction norm is supposed to describe the phenotypic plasticity of age and size at maturation in response to environmental factors. In particular, age and size at maturation are strongly determined by somatic growth rate, which in turn largely depends on environmental conditions such as food resources or temperature. In the context of harvested fish populations, earlier maturation has often been explained by a compensatory response of density-dependent growth to reduction in population size: the resulting release of trophic competition improves growth, which enables individuals to mature at earlier ages (Reznick, 1993). The reaction norm is supposed to describe those changes in maturation that are not due to variation in growth; they might be ascribable to a number of factors.

Environmental factors may induce maturation plasticity directly, without being channelled through growth. As shown for several fish species (Marteinsdottir and Begg, 2002; Grift *et al*., 2007; Mollet *et al*., 2007) individuals with better body condition have a stronger propensity to mature, irrespective of their age and size and thus growth. Ecological factors enhancing body condition, such as food abundance (Siems and Sikes, 1998) are therefore alternative causes of trends in maturation age and size. Increased temperature is also known to accelerate maturation independently from growth (Law, 2000; Grift *et al*. 2003; Kraak, 2007). In some fish species, the social structure and size composition can also influence maturation (Hutchings *et al*., 1999).

Thus, when interpreting changes in size at maturation, the evolution of maturation propensity in response to fisheries-induced selection is far from being the only possible explanation. Obviously, when mortality increases, earlier maturation is expected to evolve and counter-balance the loss of fitness due to the decrease in survival by an increase in lifelong fecundity (Gårdmark and Dieckmann, 2006; Stearns, 1992; Ernande *et al*., 2004). But, as outlined above, changes in the maturation reaction norm are ascribable to a wide number of factors, and convincing, direct evidence of the determining influence of one among these factors is generally difficult to provide. In the fisheries literature, changes in maturation propensity are ascribed to fishing-induced evolution after a thorough investigation of all other potential factors mentioned above; if their cumulated influence does not suffice to explain the observed change. However, this is rather indirect evidence, and other yet unknown factors may intervene. Therefore the debate about whether changes in maturation can be ascribed to fishing is ongoing (Marshal and McAdam, 2007). Size at maturation of exploited fish species is obviously not an indicator of the genetic effects of fishing; and is even not a specific indicator of fishing impacts on target populations.

Although maturation probability is expected to be independent of the variations in growth and mortality that confound descriptions of change based on maturity ogives (Wright *et al*., 2011) WGECO recommend that simpler indicators of the phenotypic changes in life history such as provided through the maturity ogives (which at least have the advantage that they are easily calculated and less reliant on large sample sizes) be reconsidered.

4.6 Fishing pressure based on VMS

In this section we will explore and further develop the pressure indicators for trawling impact on the different marine habitats put forward by (CEC, 2008):

- Distribution of fishing activities;
- Aggregation of fishing activities;
- Areas not impacted by mobile bottom gears.

As it is recommended to report on these indicators in conjunction, the indicators will be considered together.

The calculation of the indicators is still hampered by the fact that there are confidentiality issues that prevent access to the international VMS data which would allow the calculation of regional indicator values. Therefore the values presented in this paper are by necessity based on case studies using national data only: one using Dutch data in the Dutch EEZ, the other using Italian data in two Geographical subareas (GSAs). The two case studies have approached this independently, applying different frameworks for calculation. The methods to calculate the indicators in the case studies, however, are intended to be generic (but based on VMS data as they are collected for the DCF) and should therefore be applicable to all European waters for which this information is collected and available.

4.6.1 Method of calculation

The method of calculation is provided in CEC 2008 and additional background information is in WGECO 2009. For further detail we refer to the two case studies below.

4.6.2 Calculation: Dutch case study

In this case study we will explore and further develop the pressure indicators for trawling impact on the different marine habitats put forward by (CEC, 2008). This involves a consideration of spatial and temporal scale according to (Piet and Quirijns, 2009) in conjunction with the reconstruction of trawl tracks based on the cubic Hermite spline interpolation technique according to (Hintzen *et al.*, 2010). Finally we will evaluate their usefulness to inform EBM toward policy targets on fishing pressure as required for the CFP and seafloor integrity as expressed in the MSFD.

In this case study we calculate specific indicators for each of the DCF indicators:

1) Distribution of fishing activity;
   1.1) Total surface area trawled
   1.2) Proportion of surface area trawled
2) Aggregation of fishing activity;
   2.1) Proportion of surface area fished by specific proportion of effort
   2.2) Proportion of surface area fished at specific trawling intensity
3) Areas not impacted by mobile bottom gears;
   3.1) Cumulative proportion of surface area not impacted over a specific time period
   3.2) Proportion of surface area not impacted incorporating uncertainty.

4.6.2.1 Method of calculation

For this case study only vessels fishing with bottom gear (see Table 4.6.2.1 for an overview of gear types considered) were included. The data consisted of information on the vessels' position, speed and heading recorded approximately every two hours. Logbook information, detailing catch composition, vessel and gear characteristics and fishing effort (e.g. days at sea) information is available and is linked to the VMS data on a trip-by-trip basis. In addition, we also used detailed information on fishing behaviour and gear characteristics of these fisheries (Rijnsdorp *et al.*, 1998; Piet *et al.*, 2007; Hintzen *et al.*, 2010). All aspects of preliminary data preparation, and the calculation of the pressure indices were done using the VMStools package (Hintzen *et al.*, 2012) and sp package (Bivand *et al.*, 2008) which are available as add-on packages to the R statistical software (R Development Core Team 2011).

Table 4.6.2.1. Overview of gears used within the analyses and the percentage of registrations for each gear type.

Gear description	%
Beam trawl >300 hp, 2 * 12 meter beam	57.4
Beam trawl <300 hp, 2 * 8 meter beam	31.2
Otter trawl, 87 meter wing spread	9.8
Pair trawl, 2 * 87 meter wing spread	1.6

Both VMS and logbook data are routinely imported and stored in a central database in the Netherlands, where they are screened for consistency. Prior to analysis, both the VMS and logbook data are further checked for invalid records with the *VMStools* software which applies the following rules:

1) No latitude and longitude positions be outside northern Europe;

2) No headings (compass bearings) outside the range 0–360°;

3) No vessel speeds records exceeding 20 knots (assumed maximum steaming speed of trawlers);

4) No VMS registrations in harbours or on land;

5) Delete suspicious VMS pseudo-duplicate records (records within five minutes of a previous registration, associated with malfunction of the VMS transponder);

6) Within the logbook data departure cannot occur before the return dates.

After this data screening process the VMS and logbook data are linked to obtain the gear definitions associated with each VMS position. The identification of fishing activity for each vessel, at each VMS position, was calculated using the segmented regression approach (Bastardie *et al*., 2010) and hence only VMS registrations classified as "fishing" were used in the calculations of fishing pressure. It is known that the resolution of spatial grids is important in such analyses (Piet and Quirijns, 2009) and therefore we constructed grids at different resolutions to compare and contrast the impact of different grid resolutions on the outcome of the analyses: a 'low' resolution grid (0.6 minutes longitude by 0.3 minutes latitude, approx. cells of 600 x 600 meters), and 'high' resolution grid (0.06 minutes longitude by 0.03 minutes latitude, approximately cells of 60 x 60 meters) [Note: For all the spatial analyses a perfect sphere was assumed and, therefore, these cannot be considered 'true' GIS calculations].

As mentioned above the VMS position registrations at two hourly intervals can only realistically provide a snapshot of fishing activity, while the pressure indicators should reflect the "true" impact on the seafloor. A necessary next first step is then to reconstruct the actual fishing tracks from the two hour position registrations, and this was done using the interpolation method developed by (Hintzen *et al*., 2010). This method combined the information from the actual position of each vessel with its known compass bearing to produce the best reconstructed vessel tracks possible given the information available. Note, however that the interpolation method as it is currently available is tuned to the Dutch beam trawl fishery for which data with a much shorter interval (12 minutes) were available allowing the validation of the reconstructed trawl tracks. It needs to be assessed whether the parameters for the interpolation method also apply as well to other fisheries.

Below we present for each of the specific indicators put forward the methodology required, our results and some discussion.

Total surface area trawled

The total area trawled within each spatial grid cell was calculated based on the width of the gear, a vessel's speed and time spent in that cell. Each VMS registration is allocated to one spatial grid cell. The average time difference between the preceding and succeeding registration is taken as the time spent within the grid cell. Multiplying time spent by gear width and speed provides information on the actual trawl track (km^2) within the spatial grid cell. Aggregating all tracks within a spatial grid cell gives the total surface trawled within the specific grid cell. Aggregating over all grid cells gives the total surface area trawled. This can also be expressed as a proportion of a marine region (e.g. North Sea), habitat (e.g. muddy sand) or as we've shown here, the proportion of the Dutch EEZ. Calculations were compared by basing them on VMS registrations only as well as interpolated tracks between VMS registrations (see Figure 4.6.2.1a).

The total surface area trawled in the Dutch EEZ increases strongly from the start of the time-series to 2004, after which it decreases again (Figure 4.6.2.1a). Overall a small increase over time is observed. Interestingly, the figure also shows how the method and assumptions used in the calculations affect the outcome where the low spatial grid resolution gives markedly lower values than the high resolution grid. The use of single registrations as opposed to interpolated tracks had no effect.

This indicator shows markedly different patterns; the pattern in the total surface area trawled shows a strong increase until 2004 after which it decreases again. This can be attributed to the combined impact of two main processes. Firstly the size range of EU vessels, for which VMS transponders was mandatory, changed from >24 m to >18 m in September 2003 and subsequently to >15 m from January 1st 2005 (Piet *et al*., 2007), and secondly a decrease in overall trawling activity occurred due to the combined effects of decommissioning and increasing fuel costs (see also Abernethy *et al*., 2010).

Proportion of surface area trawled

The proportion of the area trawled is calculated by counting each grid cell that is trawled as a trawled grid cell without any consideration of how much of the grid cell is actually trawled. Aggregation over all grid cells in an area now gives the total proportion of that area trawled.

Similar to the total surface area, the proportion of area trawled (Figure 4.6.2.1b) also displayed a change over time (albeit different) but was affected even more by the method of calculation in that now the choice between single registrations or interpolated tracks also had a marked effect. While there appears to be a modest increase over time by approximately 15% at low resolution, and considerably less at high resolution, there are major differences of several orders of magnitude, depending on the method of calculation. Estimates of the proportion of the area trawled based on registrations at high spatial resolution suggest that less than 1% is fished while interpolated tracks at low spatial resolution show approximately 76% of the area is fished.

Despite the large fluctuations in total area trawled there is a gradual increase in terms of the proportion of the Dutch EEZ that is trawled, showing that previously untrawled areas are now being exploited. Note that, in theory, the total surface area trawled could exceed that of the surface area of the EEZ, while the proportion of the Dutch EEZ fished remains well below 100% due to the fact that many spatial units of the EEZ are fished several times per year.

a b

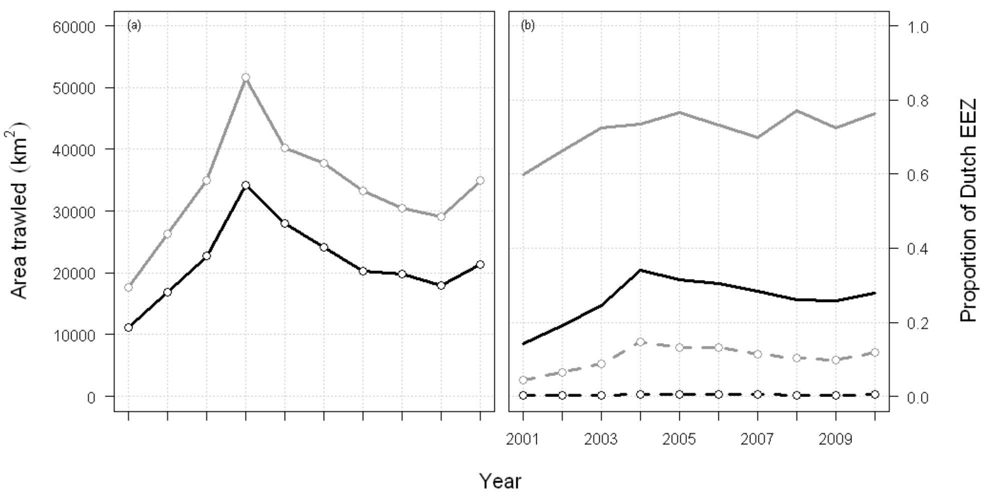


Figure 4.6.2.1. Total surface area trawled in km^2.yr^{-1} (a) and Proportion of the EEZ trawled (b) over time. The solid lines are based on interpolated trawl tracks and the dashed lines and open circles are based on single VMS registrations only. Both are calculated at two spatial grid resolutions (black=high, grey=low).

Proportion of surface area fished by specific proportion of effort

This indicator is then calculated from the previous indicator through summation of the grid cells in decreasing order until a specific percentage of the total effort (i.e. 90%) is reached. The indicator equals the total surface area of these grid cells as a proportion of the total surface area.

This indicator shows an almost identical pattern to the previous indicator but, depending on the chosen proportion of effort, at a fixed ratio of that indicator. At the same configuration where 100% of the fishing effort resulted in 76% of the area fished, 95%-90%-85% of the effort now result in respectively 40%-30%-25% of the area fished (Figure 4.6.2.2).

The value of this indicator is a fixed ratio of the "proportion of surface area trawled" (see 4.6.2.1b) and thus does not provide any additional information to that indicator. The observed reduction of approximately 50% surface area for a 10% reduction of fishing activity in the infrequently fished margins agrees with the observations of (Jennings and Lee, 2012).

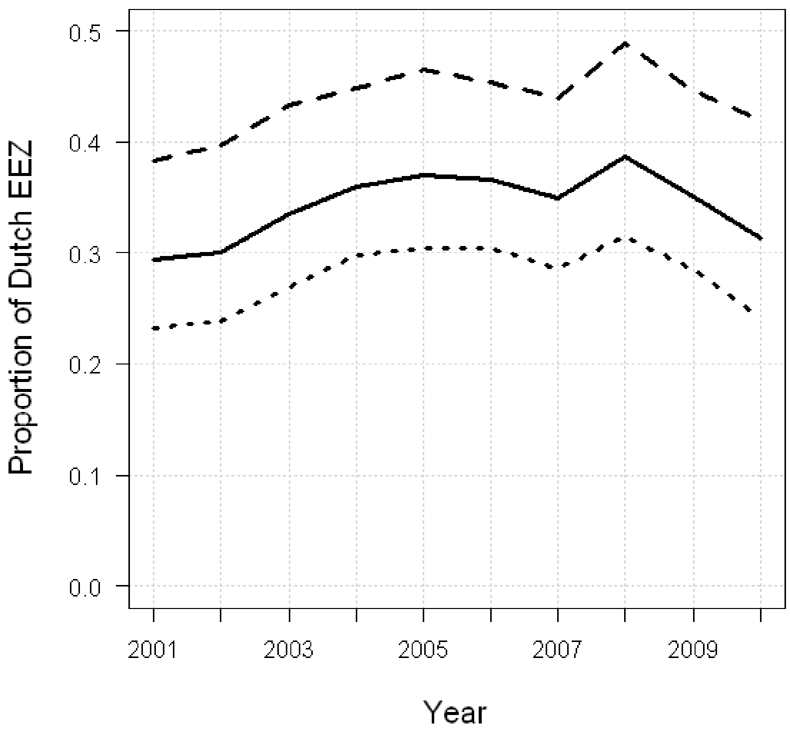


Figure 4.6.2.2. Proportion of the surface area fished by 95% (upper dashed line), 90% (solid line) and 85% (lower dashed line) of the effort.

Proportion of surface area fished at specific trawling intensity

Based on the calculations above, we can derive the intensity of trawling for each of the spatial grid cells. If the area trawled within a spatial grid cell equals its total surface, trawling intensity equals 1. Similarly if the area trawled is only half of the total surface, trawling intensity equals 0.5. The trawling intensity is calculated by aggregating over all grid cells, and given as the proportion of the total area trawled x times. As values of x have a continuous character, 'bins' have been defined for x. Proportional trawl intensity is calculated for intensities < 0.1 year^{-1}, 0.1–0.5 year^{-1}, 0.5–1 year^{-1}, 1–2 year^{-1}, 2–5 times year^{-1}, 5–10 year^{-1}, ≥10 year^{-1}.

The proportion of the Dutch EEZ that is trawled, is not trawled uniformly (Figure 4.6.2.3). Most of it (approximately 90%) is hardly trawled (frequency<0.1 year^{-1}) while about 9% is trawled with a frequency of 1-5 year^{-1} leaving only 1% to be trawled more than 5 year^{-1}. The pattern over time is similar but inverse to that of Figure 4.6.2.1b.

While the main criterion for the impact of fishing on an area, or the quality of that area, is determined by whether or not it is fished, the intensity at which it was fished does make a difference. Obviously higher intensities result in a more impacted habitat with lower quality. A direct mortality of, e.g. 20% for *Arctica islandica* based on a single passage of a gear (expressed as % of the initial density in the trawl track) (Bergman and van Santbrink, 2000) results in an annual mortality of almost 70% in an area that is trawled on average 5 year^{-1}. Few populations can withstand such annual mortalities although they could easily endure the mortality caused by the single passage of the gear.

The proportion of the surface area not fished in a particular year is the inverse of the previous indicator "*proportion of surface area trawled by year*" and it would, therefore, seem to be rather uninformative to present them together. However, the "*Cumulative proportion of surface area not impacted over specific time period*" does provide additional information that may be very relevant to describing the impact of fishing on the seafloor, or the status of the seafloor. This is because the impact of fishing on, most notably, the longer-living benthic species is determined by the proportion of an area not impacted over a longer time period.

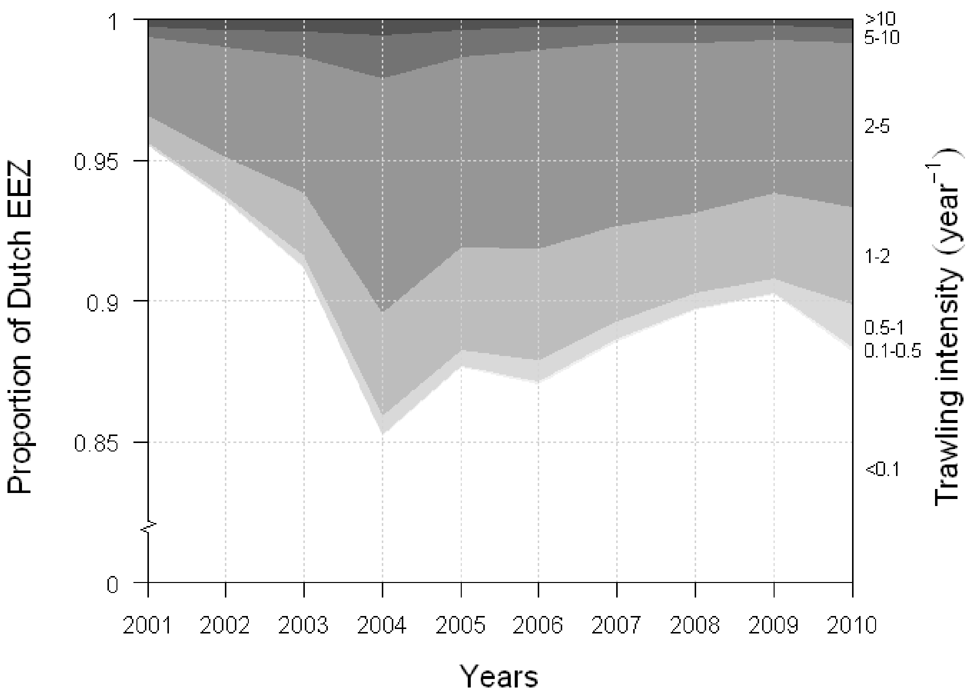


Figure 4.6.2.3. Proportion of the Dutch EEZ fished at a specific trawling intensity over time (note the y-axis break). The right y-axis indicates the intensity bins where <0.1 means that a certain proportion of the EEZ is trawled less than 0.1 times per year.

Cumulative proportion of surface area not impacted over specific time period

While the former indicator of fishing impact is most relevant to habitats with a relatively short recovery time (< 1 year), the cumulative proportion of surface area not trawled is most appropriate to habitats with longer recovery times often occurring in lower energy environments. This indicator is, therefore, calculated by adding the registrations/tracks for each additional year of fishing to those of the previous year(s). The surface area of each grid cell that has not been fished is thus integrated over successive years. The total surface unfished can then be divided by the total surface area of the EEZ. Starting in 2001, we can track how many of these spatial grid cells remain unfished in subsequent years (i.e. 2002, 2003,… until 2010, see Figure 4.6.2.4).

In each specific year the proportion of the surface area not impacted (Figure 4.6.2.4) is the inverse of what is shown in Figure 4.6.2.1b. However, over longer periods of time we see the proportion of the EEZ not being impacted by mobile bottom gears decreasing markedly. Again the method applied affects the indicator value calculated.

When based on VMS registrations at low spatial resolution in 2001 it has a value of approximately 86% and decreases to about 28% when accumulated over a ten year period. In contrast, a high resolution grid and use of interpolated tracks in the calculations shows a decrease from about 95% in 2001 to approximately 48% when fishing accumulates over the following ten years.

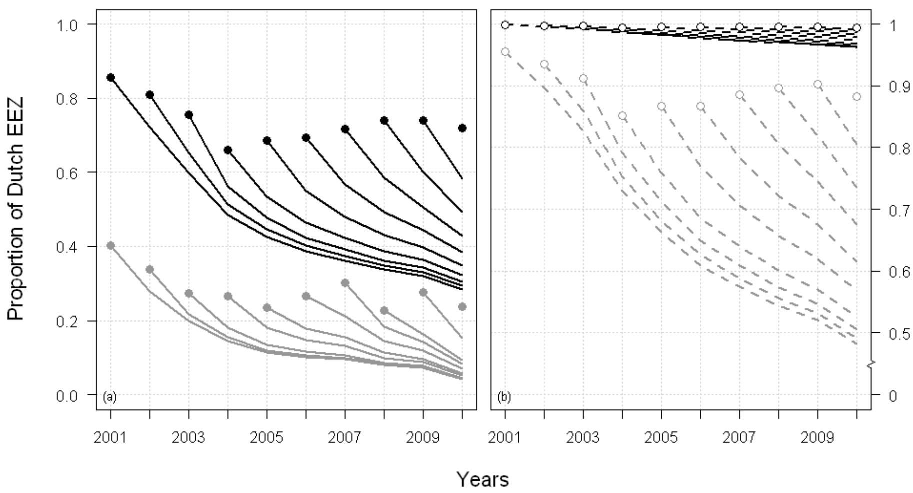


Figure 4.6.2.4. Cumulative proportion of surface area *not impacted* over a specific time period. The solid lines and points are based on single VMS registrations only (a) and the dashed lines and open points are based on interpolated trawl tracks (b). Both are calculated at two spatial grid resolutions (black=high, grey=low). Points represent the year-estimate of proportion unfished. The lines represent the cumulative proportion unfished over time.

Proportion of surface area not impacted incorporating uncertainty

This indicator incorporates the uncertainty in the estimated trawl path based on the VMS registrations and using the available interpolation techniques (Hintzen *et al.*, 2010). This uncertainty is incorporated in the estimated area impacted by fishing through the calculation of the chance of no-impact using the Confidence Interval (CI) calculation as proposed in (Hintzen *et al.*, 2010).

For any indicator that may be used to inform the decision-making process, transparency with regard to uncertainty is becoming increasingly important. An advantage of the interpolation techniques of (Hintzen *et al.*, 2010) is that they can provide a confidence interval around the estimated trawl track, allowing the consideration of uncertainty when communicating the absolute value of the indicator. The example of the Dutch EEZ shows that when only the trend is considered, incorporation of the uncertainty makes no difference and can be ignored (Figure 4.6.2.5). However, the absolute value of the indicator does depend markedly on the level of uncertainty chosen. The actual proportion of surface area not impacted varies from only 2% at the end of the period with 100% confidence (i.e. no uncertainty) to approximately 77% when confidence is lower and 40% uncertainty is allowed. This absolute value is relevant because with the current knowledge, pressure indicators, with some arbitrary percentage of the proportion of area unaffected as a policy target, are better to use than any of the available (if any) state indicators for which no reference levels exist (Rice *et al.*, 2012). The pressure indicators calculated in this study require some consideration of the uncertainty at which the indicator value is estimated and reported.

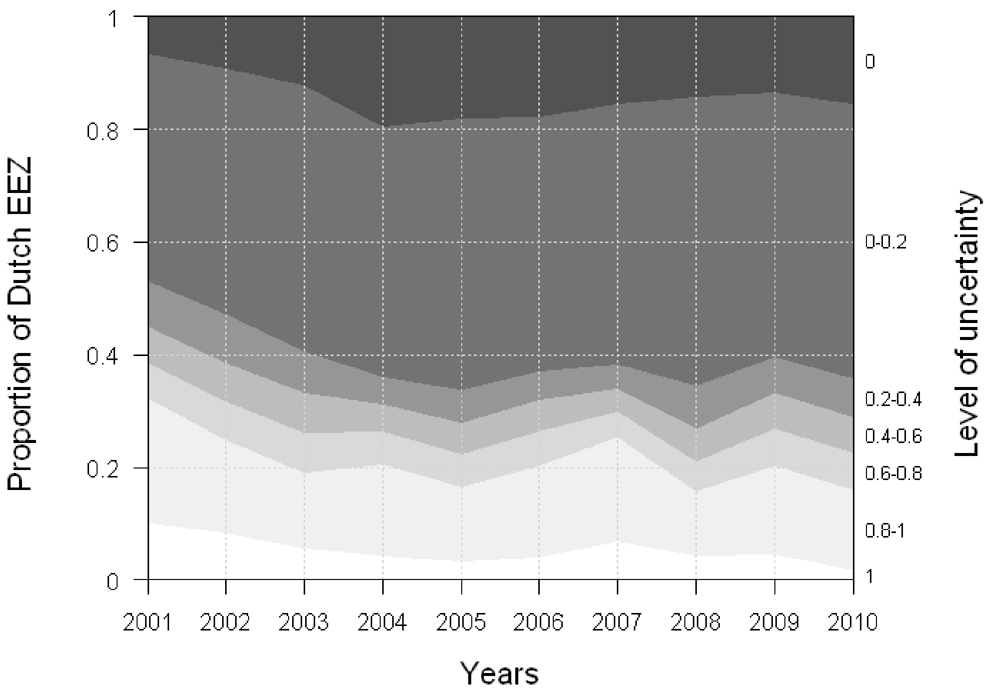


Figure 4.6.2.5. Proportion of the EEZ unfished by mobile bottom gears allowing different levels of uncertainty.

4.6.2.2 Discussion

When selecting a suite of indicators to describe the impact of fishing on the seafloor or the status of the seafloor as determined by the part not affected by fishing, several considerations apply. The overall impact of a particular fishing activity is determined by its extent, in terms of area covered by at least one passing of the gear, as well as its intensity because each additional passage of the gear may increase the impact on the habitat. Therefore, in order to describe the pressure of these activities on the habitat with one or more indicators, they must capture both of these components of impact.

Below we discuss the most appropriate method (reflected by the specific indicators) for each of the DCF indicators as they were phrased in CEC (2008) both in relation to fishing pressure but also seafloor integrity as specified in the MSFD.

DCF indicator 5 "Distribution of fishing activity" and its specific indicators developed in this study describes the fishing pressure on the ecosystem in terms of its spatial distribution and may apply to any segment of the fleet (i.e. not necessarily those segments disturbing the seafloor). These specific indicators differ in terms of their suitability as indicators of fishing pressure. The "*Total surface area trawled*" (5a) only reflects the potential of the fishing fleet to disturb specific ecosystem components, but does not provide information on the actual impact since, theoretically, all the pressure can be exerted on one small patch of the ecosystem leaving the remainder undisturbed. This critique does not apply to the "*Proportion of the surface area trawled*" (5b) which does reflect how much of an area (and the biota living there) is disturbed. This makes this indicator the preferred indicator for fishing pressure when not related to seafloor integrity.

DCF indicator 2 "Aggregation of fishing activity" is relevant as it determines the level of disturbance, predominantly the benthic community. This therefore applies mostly to those segments of the fleet that disturb the seafloor but could apply to segments that disturb other components (e.g. seabirds or marine mammals) as well. As the precise level of disturbance is less relevant to the components other than seafloor the "Proportion of surface area fished by specific proportion of effort" (6a) can be reported in conjunction with the indicator 5b as was initially intended in (CEC, 2008) although 6a provides nothing more than a fixed ratio of 5b. However, as a pressure indicator reflecting seafloor disturbance the indicator "Proportion of surface area fished at specific trawling intensity" (6b) is more appropriate than the initially proposed 2a. For the "specific intensity" level required for this indicator we propose ≥1 year^{-1} as this can be easily understood and interpreted thereby fulfilling one of the criteria for indicator selection (Rice and Rochet, 2005) unless it is possible to choose a more informed level with the knowledge available. For consistency and because DCF indicator 6 is mostly intended to reflect the disturbance to the seafloor we propose to use indicator 6b to reflect the DCF indicator 6.

DCF indicator 7 "Areas not impacted by mobile bottom gears" could be used as an indicator of fishing pressure but is probably more appropriate as a state indicator of habitat status or seafloor integrity. As habitat status is determined by the level of disturbance any indication of the intensity with which (a proportion of) the habitat is fished gives some indication of state. However, unless detailed knowledge of the relationship between trawling frequency and habitat status together with some concept of GES for seafloor integrity is available, the only useful criterion in relation to GES is the proportion of the habitat unfished (which in practice implies unfished for a sufficiently long period) as this, by definition, determines the habitat to be in GES. Note, this does not suggest that a habitat of which a certain proportion was fished at a certain (low) intensity cannot be in GES.

Results in this study have shown that calculation of the DCF indicator 7 requires two decisions that determine to a considerable extent the qualification "not impacted": (1) the time period during which the area should not be fished and (2) the level of uncertainty allowed. The first is required because the status of habitat is determined by the accumulation of fishing over a longer period, leaving a smaller proportion of the area "not impacted" which is actually interpreted as "not impacted for a period long enough to allow recovery of those characteristics that determine the quality of the habitat". The length of that period may differ depending on the type of habitat. The "*Cumulative proportion of surface area not impacted over a specific time period*" (7a) takes the accumulation of fishing into account and is, therefore, the most appropriate indicator for the status of the habitat and thus the seafloor integrity descriptor part of the MSFD. The second requirement involves the degree of uncertainty when assessing if an area is not impacted as this determines the actual value of the indicator and 100% confidence may not be the level desired by decision-makers. Moreover, different levels of confidence could be considered depending on the vulnerability of the habitats, e.g. a much higher confidence level would be required when reporting on a sensitive biogenic reef as opposed to a sandy habitat.

In this case study different methods were applied to calculate the DCF indicators required to inform decision-makers on the pressure of fishing as well as the status of the seafloor (CEC, 2008) and this also determines the choice of specific indicators. For fishing pressure DCF indicators 5 and 6 are most appropriate with "*Proportion of the surface area trawled*" (5b) as the preferred specific indicators for the former and "Proportion of surface area fished at specific trawling intensity" (6b) as the preferred specific indicators for the latter. Specifically for the métiers that disturb the bottom DCF indicator 7 is relevant both as an indicator for fishing pressure as well as a state indicator for habitat status or seafloor integrity. But in order to ascertain optimal use of these indicators in this context, agreement on the method of calculation is required. This must explicitly include (1) the spatial resolution of the grid cell framework used where higher resolutions provide more accurate estimates, (2) a choice between using position registrations only or interpolated tracks where the latter provides the most accurate values, (3) the appropriate period of time without fishing impact required for a specific habitat to recover, where longer periods are required for more sensitive habitats and (4) the level of uncertainty allowed when reporting on the proportion of an area not impacted where a higher degree of certainty will result in a smaller proportion of an area un-impacted.

4.6.3 Calculation: Italian case study

This presents the Italian experience computing the DCF indicators of fishing pressure 5-*Extension of fishing activities* and 6-*Aggregation of fishing activities*.

All the methodological comments are referred to the procedure reported in the 2009 WGECO Report, which still represents the most detailed document in this framework.

The procedure reported in the 2009 WGECO Report is quite detailed and exhaustive, under the assumption that fishing set position per month, disaggregated at the level 6 métiers, are available.

Considering that the specification of the Indicators in Appendix XIII of the DCR identifies a 3 km x 3 km grid size as optimal for representing fleet distributions, the computation of indicator 5 is very easy: it is sufficient to plot fishing set position on the grid and then count the number of cells with at least one point. The value of indicator is then determined by multiplying the number of cells for 9 km^2. Thus, the expression of the indicator 5 is:

$$E_{m,a} = n_{m,a} \times 9$$

Where $E_{m,a}$ is the value (in km^2) of the indicator at month m, for métier a, and nm,a is the number of grid cells "activated" (with at least one point).

The indicator 6 represents the minimal area in which falls the 90% of the total number of fishing points recorded in a given month. This can be computed by sorting, in a decreasing order, cells by fishing points and then cutting the series when the cumulated number of fishing points reaches the 90% of the total value. The expression of the indicator 6 is:

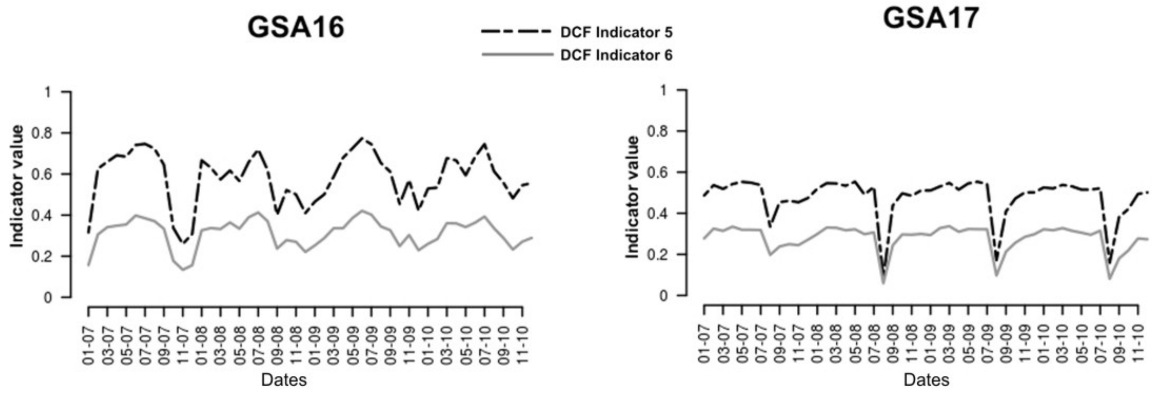
$$Am, a = n90_{m,a} \times 9$$

Where $A_{m,a}$ is the value (in km^2) of the indicator at month m, for métier a, and n90,a is the number of grid cells summing up the 90% of the total number of fishing points.

Following this procedure, the complete time-series for DCF indicators 5 and 6 have been computed and inspected for the 7 Geographical subareas in which the Italian seas are partitioned for data collection and management purposes.

Relevant results are reported for two case study GSAs, namely GSA16 (Strait of Sicily) and GSA17 (North of Adriatic Sea).

The following plots represent the pattern obtained for the two DCF indicators in the 2007–2010 period.



A first result is that, in all cases, the behaviour of indicator 6 closely follows that of indicator 5, up to an additive constant. The values also show important fluctuations, which seem to be seasonal with a yearly frequency. Within each year, the maximum effort in terms of exploited area is deployed between March and August, while a temporal stop in September is followed by a slow reprise of the activity. In general, both indicators seem to be fairly stationary during the period 2007–2010.

General issues (common to several indicators) and their possible solutions

- The original versions of the three indicators are not formulated as ratios, but just as raw measures of extension (i.e. they are areas). In this way, the values computed for different regions or seas are not comparable. A possible solution is to divide the value of the indicator by the total area of the inspected region, or even better by the theoretical exploitable area (e.g. the continental shelf for métiers such as trawl). This would allow obtaining standardized value, ranging between 0 and 1. This approach has been already applied for the Italian seas, leading to the results reported above.
- Some steps of VMS data processing require the setting of appropriate parameters, such as the frequency of pings (either native or interpolated) and the size of cell grids. Lambert *et al*. (2012) corroborated that these parameters influence significantly the indicators and can lead to an underestimation of fishing impact on epifaunal communities. A possible solution is to calibrate these parameters by an analytical procedure. However, in order to prevent loss of comparability across different regions, a set of reasonable values should be selected by comparing different areas and environments.
- Following the previous point, particular attention should be put on the frequency of pings. In general, VMS signals are provided at a low frequency (0.5–1 per hour), and different methods exist to interpolate and obtain high frequency dataset (Hintzen *et al*., 2010; Russo *et al*., 2011a). Unfortunately, each method is characterized by associated errors which can be negligible or not, depending on the situation. The drastic solution of increase pings rate to very high values (e.g. 5–10 pings per hour) is not applicable due to the costs. Lambert *et al*. (2012) suggested an optimal compromise of 2 pings per hour. This value also arose from other ICES context (Workshop on the utility of commercial cpue and VMS data in assessment WKCPUEFFORT) and seems to be a reasonable solution. How-

ever, a rate of 5 pings per hour (1 each 20 minutes) does not represent the final solution to this problem, since some very important fishing activities (e.g. pair trawl in the Mediterranean) are characterized by a short temporal duration of hauls (20 minutes). At present, for the computation of DCF indicators in the Italian seas, a frequency of ten minutes is used. This frequency is obtained after interpolation of native VMS pings using the method described in Russo *et al.*, 2011a.

- The primary source of information for the computation of these indicators is represented by processed VMS data, which ultimately provide a detailed spatial localization of fishing effort in terms of fishing points (sets). However, there is a general agreement about the fact that some processing steps determine the presence of false positive/negative. This problem contributes, among others, to generate noise in the overall distribution. The present formulation of indicators, and particularly of indicator 5, seems to be very sensitive to this noise (Russo *et al.*, under review). Some procedure should be identified to clean the dataset and minimize these errors. A rough solution could be reached by ruling out cells with very low value of fishing points (e.g. 1 per month). More sophisticated approaches could involve the analysis of spatio-temporal patterns of fishing effort deployment (see forward in this document).
- Present formulation of DCF indicators establish that each indicator must be computed with respect to a level of disaggregation of fishing activity that corresponds to métiers 6. Data Collection Regulations and the Data Collection Framework for the Common Fishery Policy (EC, 2008a,b), identified, for the fishing activity in the Mediterranean and Black Seas, a list of 28 métiers (see Table 1 of ICES, 2009b) of level 5/6. Considering the complexity of Mediterranean fishing activities, this seems to be critical points and a serious bottleneck for the analysis. In fact, disaggregation of fishing activity to this level of métiers requires data that are often unavailable or unreliable (e.g. logbooks). While new approaches have been developed to overcome this limitation (Russo *et al.*, 2011b), it is reasonable that some error is still present and affects the calculation of the indicators. A possible and reasonable solution can be identified in the disaggregation of fishing activity at a higher level (e.g. Métier level 4). This is also justifiable considering that Métier of level 4 corresponds to gear deployed, and that a finer disaggregation with respect to targeted communities or assemblages does not introduce addition information about environmental disturbance.

Specific issues

- **Indicator 6:** it seems that the present formulation of this indicator is arbitrary, since the threshold of 90% in grouping of fishing points is not justified by evidences (Fock, 2008; ICES, 2009). More robust thresholds should be identified. In alternative, other metrics could be selected to capture the information provided by this indicator (that is the level at which the fishing activity is concentrated in certain areas). One of this is represented by the largely known Gini's index (Russo *et al.*, under review).
- **Indicator 6:** the previous suggestion is reinforced by the preliminary observation (Russo *et al.*, under review) that the present formulation of indicator 6 does not provide significant information, since the pattern observed

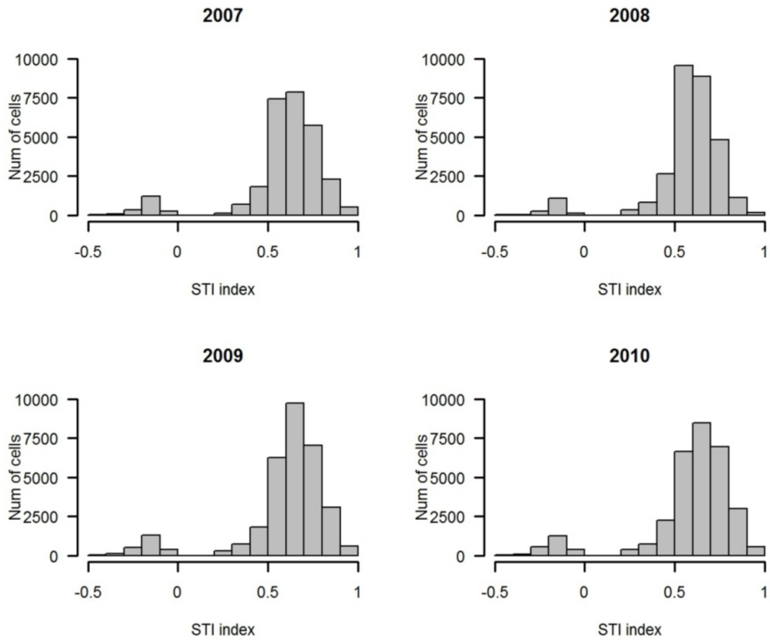
for this indicator closely follows that of indicator 5, up to an additive constant.

- **Indicator 6:** if the aim of this indicator is to capture the heterogeneity of the spatial distribution of fishing, other metrics could be applied, such as the number of spatial patches (Woillez *et al*., 2009).
- **Indicator 7:** the yearly temporal scale on which this indicator must be computed seems to be inappropriate. Different towed gears are characterized by different physical effects on the environment which, in turn, requires different time to recover after disturbance. Considering that this indicator is firstly aimed to assess the extension of areas not perturbed by towed gear, it could be useful to compute it at different rate (i.e. seasonal, half-yearly, yearly and biennal)

Proposal for alternative DCF indicators

The argumentation exposed above could be used to detect some limits, and then to suggest possible improvements for the DCF pressure indicators. A major drawback of the DCF indicators 5 and 6 is that both of them consider, as input for computation, the count grid directly obtained by plotting the fishing positions. This leads to the fact that a number of cells containing just one or few fishing points are retained throughout the analysis. As a consequence, DCF indicator 5 in particular could intuitively produce an overestimation of exploited areas. A new version of DCF indicator 5 could be computed using the output of the procedure presented Russo *et al*. (under review): this new indicator, called Fishing grounds extension (FGE) represents the total area of the fishing grounds exploited, for each GSA, for each month. The basic difference with the DCF version of this indicator is that, in this case, only cells belonging to fishing grounds were considered. More in detail, Fishing grounds (and cells belonging to them) are yearly identified by analysis of the spatio-temporal pattern of fishing effort deployment, and then selecting the group of cells for which the effort is statistically significant in space and time. In contrast, areas sporadically visited or isolated from other exploited cells are ruled out and ignored in computation of indicators.

The Griffith's spatio-temporal index (STI) of autocorrelation (Griffith, 1981; Henebry, 1995; Fortin and Dale, 2005) can be used, among others, to address this aim. Application of this index to the Italian seas, for the years 2007–2010, allowed obtaining the patterns showed in the next figure.



In all the four cases the distribution seems to be composed by two parts, the rightmost one corresponding to cells with a high value of the Griffith's STI. The homogeneity of these four distributions was tested using the Kolmogorov-Smirnoff test (R Development Core Team, 2008), which reports the expected result that we can consider the data as a whole. This suggested the existence of a conservative phenomenon underpinning the data pattern. We assumed that each of these distributions consists of two components, in which the rightmost component corresponds to fishing grounds. The other component, instead, can be explained as the result of exploration activity (i.e. searching for new fishing grounds) or noise. In this way, the identification of fishing grounds can be reduced to the decomposition of this mixture and the assignment of each cell to one of the two components.

This rationale has been used to re-compute indicator 5 (now called **Fishing ground extension**). The results obtained for the two sample GSAs are showed in the following plots, together with the corresponding trends of the Gini's index of concentration.

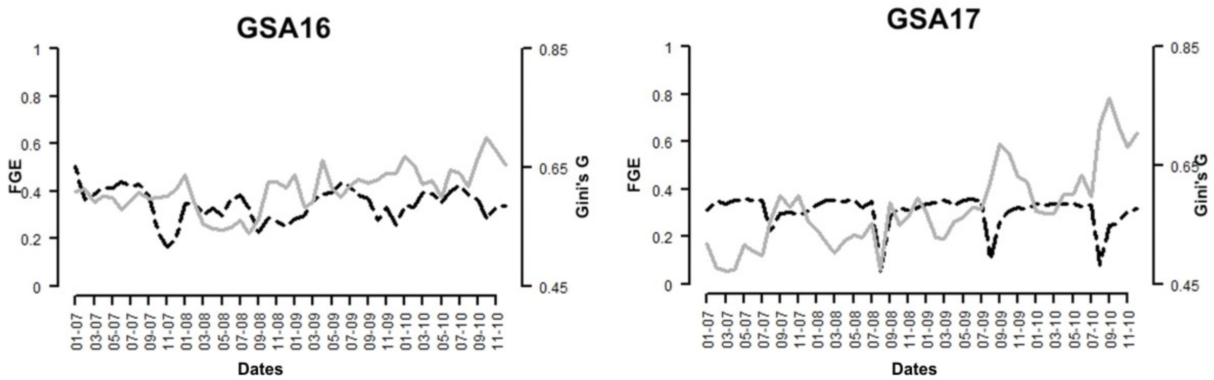


Figure 4.6.3.1. Two alternative indicators for DCF 6: Fishing ground extension (FSE, black) and Gini's index of concentration (grey).

Also in this case, seasonality is present. The patterns for extension of fishing grounds seemed to be quite stable and less variable than those described for DCF indicator 5.

The new version of indicator 6, defined by Gini's index, corroborated increasing patterns.

When considering these indicators as an alternative or improvement to DCF indicator 6 it should be realized that while an advantage of the FSE may be the reduction of noise but as it also results in the exclusion of at least some of the exploratory fishing behaviour which is probably the most important aspect that needs to be captured in such an indicator it should not be considered an improvement to the initial indicator 6.

Gini's index does appear to pick up a signal not captured by the initial indicator 6. What that signal actually represents needs to be clarified before this indicator can be applied as an alternative or even improvement to indicator 6.

Beyond computation: analysis of indicators

The time-series for ecological indicators is growing. A number of six years (2006–2011) of data are now available. For the monthly scale-computed indicators (namely pressure indicators 5 and 6) this implies that time-series are 72 points long. In this way, an approach should be developed to inspect pattern for these time-series and possibly extract emerging information in terms of trend for space use by fisheries. However, it should be stressed that these series are strongly characterized by autocorrelation and periodicity (Russo *et al*., under review).

ARMA models (Box, Jenkins, Reinsel, 2008) represent a standard tool in time-series analysis, as they can capture the autocorrelation structure which is generally found in longitudinal data. For this class of models, however, the identification and estimation steps require a large sample size, and this requirement won't be satisfied for the next years. In the meanwhile, we can drop the longitudinal perspective and assume a regression model in which our series can be decomposed in a trend and a seasonal component.

$$y_t = T_t + S_t + \varepsilon_t , \qquad \varepsilon_t \sim WN(\sigma^2)$$

where T_t can be chosen as a polynomial function of time, and the seasonal pattern can be captured by a set of dummies. Standard procedures can be carried out to select the best model (e.g. the Akaike Information criterion) and estimate their parameters (e.g. Ordinary Least squares).

4.6.4 Synthesis and recommendations

From the two case studies presented the following (often common) issues emerge:

- Data cleaning is necessary and should be done consistently following some protocol. This could be drafted from the experiences gained in various studies.
- In contrast to how the indicators were initially defined, i.e. providing some measure of extent expressed in e.g. km^2 they should be reported as a proportion to the total regional area or possibly only some relevant part of that region (see proposed Fishing grounds indicator).
- The resolution of the spatial grid should be reconsidered as it strongly determines the value of the indicator. The choice of what can be considered the most appropriate resolution, however, is linked to the interval between VMS position registrations. The initially proposed 3x3 km^2 grid is probably appropriate to the two hour interval that usually applies. Acknowledging that the resolution of the grid cells strongly affects the value of the indica-

tor with higher resolutions providing more realistic values two options emerge: an increase of the VMS frequency or applying the existing method to create the trawl track through interpolation and with some notion of uncertainty (see below). For all indicators below we propose to use the highest spatial resolution possible and base the calculation of the indicators on the interpolated track. For this at least two different interpolation methods (as applied in the two case studies) are available.

- The temporal resolution is also an issue that needs to be considered. The examples show the indicators can be calculated on a monthly or annual basis. However, certainly for the DCF indicator 7 "Areas not impacted by mobile bottom gears" indicator it was shown relevant to determine a cumulative impact over a number of years. In that case only the annual basis should be applied. Other than recurring seasonal fluctuations the monthly calculation of the three indicators did not reveal any additional information to the annual indicator values.
- The proposed calculation of the indicator per level 6 métiers is not considered realistic. We propose to calculate the indicators using level 4 métiers.
- In addition modifications to the existing indicators are suggested or alternative indicators proposed:
    - For the DCF indicator 5 "Distribution of fishing activity" we now propose the "Proportion of surface area trawled".
    - For the DCF indicator 6 "Aggregation of fishing activity" several alternative indicators are developed that at least provide more information to the previous indicator than the fixed ratio observed for the initial indicator 6. Because there are still issues with some of these indicators as to what they represent we are proposing "The Proportion of surface area fished at specific trawling intensity" as the preferred indicator. This has the added benefit that it complements the DCF indicator 7.
    - The DCF indicator 7 "Areas not impacted by mobile bottom gears" is an important indicator as it not only can be used to describe fishing pressure but also the state of certain habitats or seafloor integrity. However, as the absolute value of the indicator is relevant when using it as a state indicator issues that affect the value of the indicator (e.g. the period to derive the cumulative impact of fishing and the level of uncertainty) and that need to be considered before qualifying an area as "not impacted" need to be addressed.

A final warning when interpreting the indicators and certainly when this involves absolute values is that in spite of all the improvements in the methodology to calculate the indicators they only reflect the part of the fishing fleet equipped with VMS transponders which in some regions or for some fisheries excludes a large part of the fleet.

4.7 Discarding rates of commercially exploited species

On-board observer programmes are run by member states under the DCF scheme, which specifies sampling strata (area, time, and métier defined by gear, vessel size, target species group, and mesh size), and sets precision targets for estimates of discards amounts of long lists of species (European Union, 2008, European Union, 2010). Theoretically, this should be sufficient to estimate discarding rates for commercially exploited species. Indeed, discarding rates and/or amounts are made available by member states for the assessment of an increasing number of stocks. However, several caveats have to be raised concerning the quality of these estimates.

Even with a detailed stratification such as imposed by the DCF, obtaining precise and accurate estimates of discard amounts is difficult owing to

i) the inherently high variability of catch and discards;

ii) the limited sampling effort that member states can afford;

iii) the difficulty in observing a representative sample of trips, because many programmes operate on a voluntary basis, the presence of the observer on board modifies discarding behaviours, and fishing activities change continuously under various drivers, making it very difficult to define an appropriate sampling plan beforehand; and

iv) the generally limited quality of data used to raise discard samples, such as effort or landing data. For these reasons discard estimates are likely to be poorly estimated and should never be shown without a confidence interval (which might be wide). In principle discarding rates suffer one less source of error since no raising variable is necessary to estimate them (although they are still required to estimate the estimate precision). However, as soon as a species is caught by several métiers in a given area, raising variables are required to weight the discarding rates of each métier. If discarding rates are to be provided per species per region, it is very likely that several métiers will be involved, thereby necessitating some raising procedure and introducing the errors associated with this.

Further, sampling programmes are not standardized across countries, introducing potential differences in the data collected and the way they can be used to estimate discarding rates (ICES, 2011). Moreover, the DCF sets out precision levels but does not include any requirements about bias. Bias is introduced to sampling schemes when samples are not representative of the population. Improving the data quality by reducing bias should be prioritized over increasing precision levels. Several dedicated working groups provided recommendations for sampling and estimating discards (ICES, 2007; ICES, 2010; ICES, 2011).

Because we had no access to any discard data we did not attempt calculation of any indicators. Access to international DCF data (discarding or otherwise) often hampers the calculation of indicators and should be resolved.

4.8 Fuel efficiency of fish capture

Data are collected by the EC on fuel usage and the amount of fish caught. These have been reported in terms of "Fuel efficiency of Fish Capture" in a document produced by the JRC. http://energyefficiency-fisheries.jrc.ec.europa.eu/c/document_library/get_file?uuid=53ffb60f-a8c3-4e21-ab4a-549b56e76baf&groupId=12762.

The document presents the results in terms of an Energy Efficiency ratio; landings quantity by fuel used and as Ratio between Landing Value and Fuel Cost. The first expressing the fuel efficiency in terms of the fish biomass, and the second in terms of revenue.

Aggregated results for the entire EU fishing fleet are presented in Figure 4.8.1.

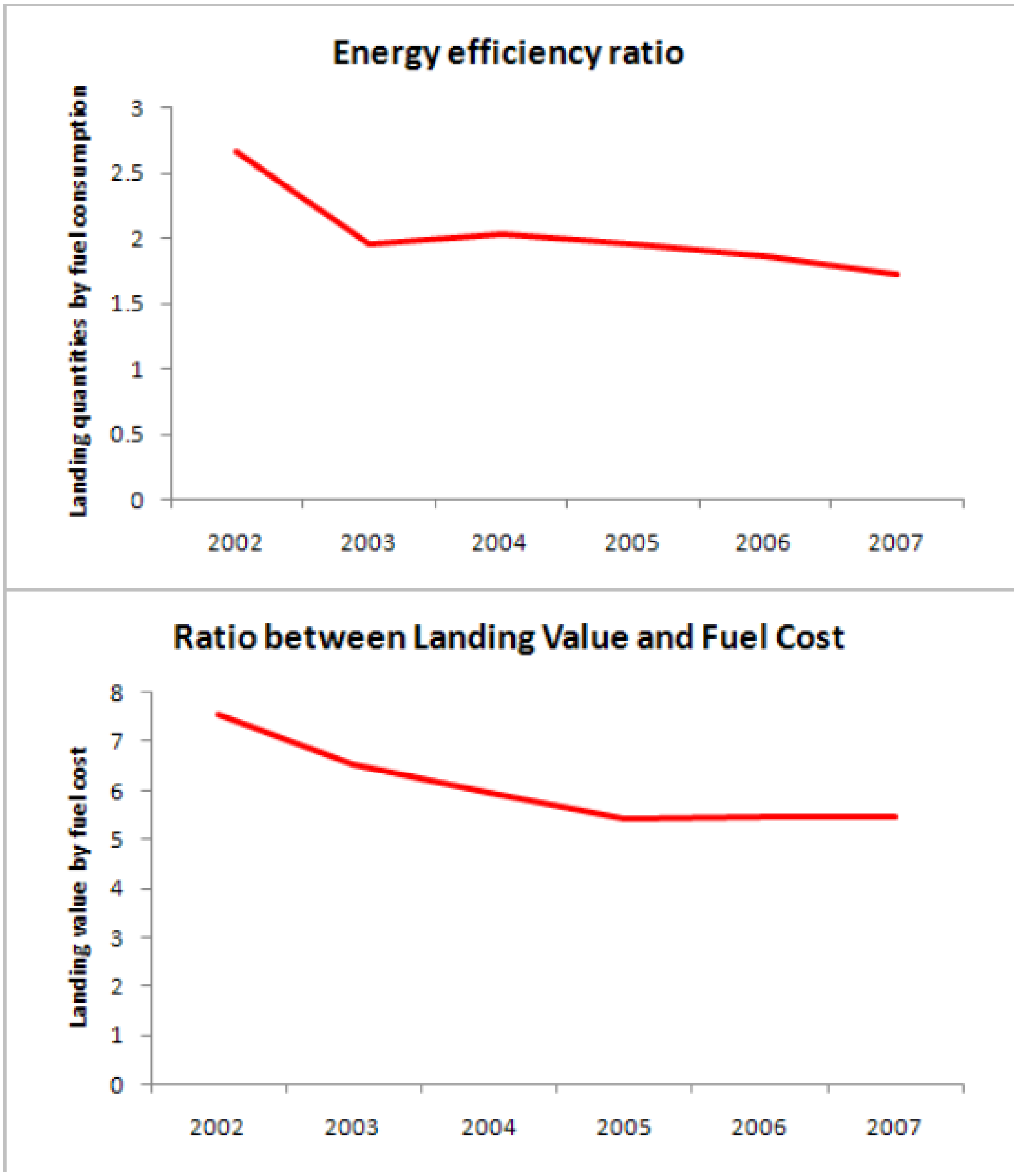


Figure 4.8.1. Top panel: Energy efficiency ratio for the fleet 2002–2007, Bottom panel: Ratio between landing value and fuel cost.

From this evidence, the information on fuel efficiency is being collected and used in an appropriate manner. WGECO does not propose any changes or re-analyses.

4.9 Link of DCF indicators to the MSFD

A summary of the potential relationship between MSFD descriptors/criteria and DCF indicators is presented in Table 4.9.1. Prior to discussion of the application of the DCF indicators it should be noted that the DCF indicators were proposed to integrate general environmental considerations into fisheries management, rather than the specific requirements of the MSFD, and were only specified with provisional reference levels or for use with reference trends, rather than specific reference levels.

Table 4.9.1. Relationship between the DCF indicators and MSFD criteria for GES.

	Indicator	Criteria	Fixed calculation method	Reference level
1	Conservation status of fish species	1.2.1	Modifications proposed. As there are several issues with the calculation of the indicator further revisions or alternative indicators should be considered	Proposed
2	Proportion of large fish	1.7.1, 4.2.1	Regionally specified threshold	Proposed in regions
3	Mean maximum length of fish	1.7.1		No
4	Size at maturation of exploited fish species	3.3.4	Modifications proposed that require less intensive sampling	No
5	Distribution of fishing activities		Improved methodology proposed and issues are identified. Data access prevents calculation of international indicators	No
6	Aggregation of fishing activities		Improved methodology proposed and issues are identified. Data access prevents calculation of international indicators	No
7	Areas not impacted by mobile bottom gears	1.6, 6.1.2	Improved methodology proposed and issues are identified. Data access prevents calculation of international indicators	No
8	Discarding rates of commercially exploited species		Data access prevents calculation of international indicators	No
9	Discarding rates in relation to landed value		Data access prevents calculation of international indicators	No
10	Fuel efficiency of fish capture			No

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5 ToRc) Ecosystem impact assessment of bottom fishing

Request from NEAFC

"Propose appropriate elements to be included in Ecological Risk Assessments of bottom fishing in the context of the proposed 2012 review of the bottom fisheries regulations implemented in the NEAFC Regulatory Area. Include appropriate justifications and background as appropriate."

5.1 Introduction

The UN General Assembly 61/105 (Article 83a) refers to the need for assessment of significant adverse impacts by regional fisheries management organizations:

"To assess, on the basis of the best available scientific information, whether individual bottom fishing activities would have significant adverse impacts on vulnerable marine ecosystems, and to ensure that if it is assessed that these activities would have significant adverse impacts, they are managed to prevent such impacts, or not authorized to proceed."

This is re-iterated in UN General Assembly 64/72 (Article 119a):

"Conduct the assessments called for in paragraph 83 (a) of resolution 61/105, consistent with the Guidelines, and ensure that vessels do not engage in bottom fishing until such assessments have been carried out."

NEAFC requested ICES to propose elements to be included in impact assessments, required to satisfy the NEAFC bottom fishing regulations in the NEAFC Regulatory Area (RA). ICES (2011a) discussed the potential applications of ecological risk assessment methods. Here we build on that previous work and examine a non-exhaustive list of (1) approaches to ecological risk assessment and (2) detect elements which are relevant to impact assessment in the NEAFC RA.

5.2 Approaches to ecological risk assessment

5.2.1 Ecological risk assessment of the effects of fishing (Hobday *et al.*, 2007, 2011)

Ecological risk assessment for the effects of fishing (ERAEF) has been applied to over 30 fisheries, mainly in Australia. The framework detailed by Hobday *et al.* (2007, 2011) entails a three level approach based on the availability of information. The first step of the assessment includes all potential effects and ecosystem components. They are evaluated in a *Scale-Intensity Consequence Analysis* (SICA), which is qualitative and intended to be a quick scan of relevant impacts to consider in the next stage, level 2. That level involves a *Productivity Susceptibility Analysis* (PSA) that uses a scoring system for evaluation. Those species that are identified as being at high risk to impact from the fishing operation from the level 2 analysis (ecosystem components at high risk) are further evaluated in a quantitative assessment (level 3). Lack of evidence is dealt with during the evolvement from level 1 to level 3, using a precautionary approach. If information is absent, the fishing activities are assumed to cause a potential high risk and are further assessed at a higher level. Both the SICA (level 1) and PSA (level 2) analyses call for explicitly identifying the level of confidence in the risk valuation so that areas can be easily identified where uncertainty is causing higher levels of risk to be indicated.

The ecosystem components that were evaluated in the ERAEF by Hobday *et al.* (2007, 2011) relate to the five focal areas under Australian environmental legislation. These are (1) target species, (2) by-product and bycatch species, (3) threatened, endangered protected (TEP) species, (4) habitats and (5) ecological communities. The interactions between the fishery and these ecosystem components are organized in six categories: (1) capture/removal, (2) direct impact without capture, (3) addition/movement of biological material, (4) addition of non-biological material, (5) disturbance of physical processes and (6) external hazards. These six interaction types are called hazards. They are defined as activities undertaken in the process of fishing which have the potential to adversely impact on ecological components.

5.2.1.1 Level 1: Scale-Intensity Consequence Analysis (SICA)

The Scale-Intensity Consequence Analysis is a rapid screening tool to identify the ecosystem components and hazards that need to be considered and those that do not. The analysis is quick, because it assesses only the worst case scenario. Therefore the unit and subcomponent of each ecosystem component that is most at risk is identified by expert judgment. There are three categories of units with components species, habitats and communities. Subcomponents are attributes, such as population size, age distribution, reproductive capacity, etc. The units and subcomponents that are most at risk are evaluated for each of the hazards. Their spatial and temporal scale is scored and their assessment is operationalized by selecting an appropriate objective.

Data requirements

There are no data requirements. The methodology entirely relies on expert judgment and can involve both stakeholders and scientists.

Strengths/weaknesses

The tool rapidly detects which impacts for each fishing activity are at a risk for each of the ecosystem components evaluated. Whereas the tool highlights the important ecosystem components and hazards, it is in itself not an impact assessment. Hence, it cannot be used to indicate whether an impact is significantly adverse, but it rules out the impacts that are not likely to be significantly adverse. As only one species is assessed per activity the selection of the species to assess is very critical and has the potential to affect results. We view the name of the approach (Scale-Intensity Consequence Analysis) as unfortunate as it suggests a rigorous evaluation and prefer the Exposure-Effects Risk Assessment terminology referred to by Hobday *et al.* (2011) or simply "initial screening" as being more reflective of the process.

Case studies

For the ecosystem component 'target species' in the western English Channel otter trawl fishery for instance, the species 'whiting' is chosen as a unit for evaluation, and its spawning–stock biomass is the subcomponent to be analysed (Cotter and Lart, 2011). The selected operational objective is a SSB > safe biological limits. Following the identification of this objective, the intensity of the fishery disturbance is scored. Locally occurring and hardly detectable impacts receive low scores, whereas widespread and continual impacts are highly scored. The consequences of the different levels of intensity are scored from 1 to 6 by evaluating whether the operational objective can be attained for a certain hazard. The end product of the SICA assessment is the identification of hazards which need to be considered in further impact assessments and those that are not relevant.

5.2.1.2 Level 2: Productivity Susceptibility Analysis (PSA)

In the Hobday *et al.* (2007, 2011) framework, the PSA is a semi-quantitative approach based on scoring each ecosystem component for the hazards identified in SICA. Productivity and Susceptibility are estimated from scoring attributes. Generally, Productivity of a species is based on life-history traits, such as longevity, reproductive strategy, fecundity, etc., which in turn determine its ability to sustain exploitation or recover after depletion. Susceptibility is a function of the level of the interaction of a particular species with a fishery. As an example, Hobday *et al.* (2011) identified four aspects of Susceptibility: Availability, Encounterability, Selectivity and Post-capture Mortality, each of which is broken down to attributes. Availability of a species is estimated from the overlap of the distribution of the fishery and the distribution of the investigated ecosystem components. Encounterability represents the physical interaction with the fishing gear. Selectivity and Post-capture Mortality are the retention of a species by the fishing gear, and its consequences. The Productivity and Susceptibility attributes are scored as 1(low), 2 (medium) or 3 (high) based on cut-off scores that are not directly linked to biological or ecological values but determined from an analysis of the overall distributions of each value followed by a division into three units reflecting low, medium and high. Missing attributes are scored as a 3. The importance of a particular attribute to a species/habitat is generally ranked, but often some attributes receive more weight than others. The final scores for Susceptibility and Productivity are plotted on a PSA plot (Figure. 5.2.1.2.1), and the risk is calculated as the Euclidean distance from the origin.

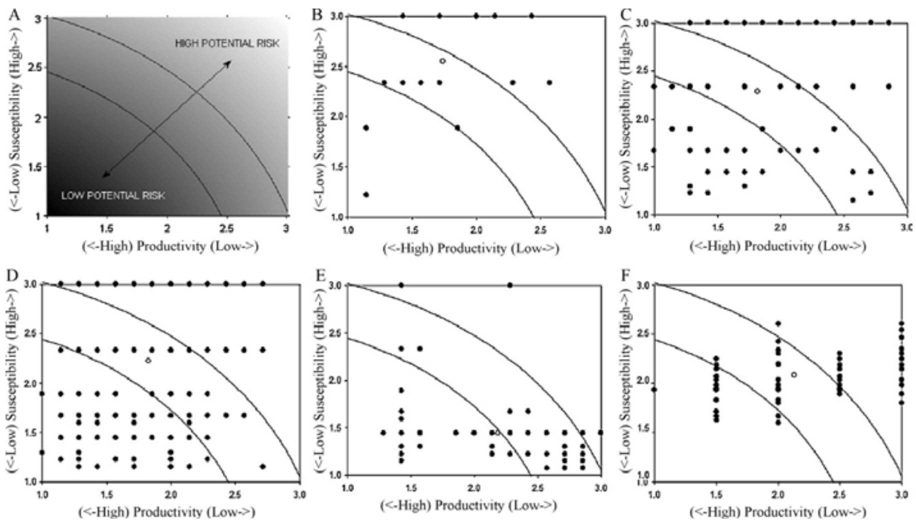

Figure 5.2.1.2.1. Productivity Susceptibility Analysis (PSA). (A) The axes on which risk to the ecological units is plotted. The x-axis score derives from attributes that influence the productivity of a unit, or its ability to recover after impact from fishing. The y-axis score derives from attributes that influence the susceptibility of the unit to impacts from fishing. The combination of susceptibility and productivity determines the relative risk to a unit, i.e. units with high susceptibility and low productivity are at highest risk, while units with low susceptibility and high productivity are at lowest risk. The curved lines divide the PSA plot into thirds, representing low, medium and high risk, and group units of similar risk levels. (B–F) Results from the Level 2 PSA analysis of the SESSF otter trawl fishery. (B) Target species, (C) by-product species, (D) bycatch species, (E) TEP species, and (F) habitat types. Note the species in the by-product/bycatch component are plotted separately here due to the large number of species. The open circle on each plot shows the mean for the component. Multiple units can be plotted at the same point if the scores are the same (from Figure 3 of Hobday *et al.*, 2011).

Data requirements

The PSA approach is semi-quantitative. Scoring of individual attributes is based on various sources of information but relies heavily on expert judgement. Stakeholder involvement is generally less than in the SICA assessment.

Strengths/weaknesses

The advantage of the PSA approach is that it is quite simple, easily repeatable and robust and may be useful in data poor areas. However, the method could be used with very few data and based mostly on expert judgment. The PSA methodology is only semi-quantitative, yet the graphical outputs present the data in a format that is associated with quantitative analyses. This has the potential to mislead and the onus is on those applying the approach to provide complete transparency of the process used to derive the table of attributes and cut-offs used to assign risk. In the Hobday *et al.* analyses (2007, 2011) the cut-offs were established by the range of the distributions and subsequent desire to have three risk categories (H, M, L). Ideally the cut-offs would be biologically based. For example for exploited fish species cut-offs which link to spawning–stock biomass (SSB) or biomass (B) reference points would provide a more rigorous underpinning to the selection of cut-offs. As with the SICA approach, the choice of attributes to assess will introduce bias into the outcomes and should be considered carefully. The PSA approach may be useful for species with the above caveats but the application to habitats and communities is less certain. Technically this can of course be achieved but what meaning the outputs have is debatable. Interactions between species and the function that they play in an ecosystem are not directly considered. For example, for cold-water coral reefs there may be a difference in the size of a habitat that is required to maintain different ecosystem functions. For example, cold-water corals produce large amounts of nitrogen-rich mucus which has been shown to locally stimulate microbial activity and may function as a vector for carbon and nutrient cycling through the microbial loop. They also provide niche space through provision of three-dimensional structure which locally increases biodiversity. Each of these ecological functions could have different susceptibilities separate from the issue of the amount required to maintain the reproductive capacity of the main species. Further, there are likely other functions that would need to be considered. PSA oversimplifies the role of habitat in ecosystems and could inadvertently present a false outcome. In general, there is a risk that this analysis may overestimate the risk of fishing to some habitat types, and underestimate the risk of fishing to others. This could be due to the fact that the assessment of risk of a species or habitat to fishing is likely to be heavily influenced on the attributes considered in the analysis and how are they weighed. Evaluation of attributes in PSA is a subjective process, and this means that the assessments of such analyses could depend very much on the selection of attributes and how they are evaluated. The fact that the scores are additive could magnify this bias. Hobday *et al.* (2001; their Figure 5) further note that level 2 methods generate a bias towards too optimistic assessments. We also have reservations over equating the Euclidean Distance from the origin with risk.

Case studies

Productivity and Susceptibility Analysis (PSA) was originally developed as an ecological risk assessment tool for the Australian prawn fishery (Milton, 2001; Stobutzki *et al.*, 2001, 2002). While most studies have evaluated the risk of fisheries on fish (Milton, 2001; Stobutzki *et al.*, 2001, 2002; Patrick *et al.*, 2010; Watling *et al.*, 2011) but there have been recent attempts to evaluate other habitat types such benthic habitats (e.g. Williams *et al.*, 2011).

Watling *et al.* (2011) evaluated the risk of the Northeast Atlantic mixed trawl fishery for 16 deep-sea fish species. Formal analytical assessment cannot be carried out for most of them, but much effort was invested in applying the best possible assessment method and the assessment for some species has been recently benchmarked (ICES, 2010). Considering these data limitations, Watling *et al.* (2011) found the PSA approach to be appropriate. Following closely the application of the PSA methodology as described in Patrick *et al*. (2010), they identified seven attributes characterizing the Productivity; maximum size and age, estimated natural mortality, measured fecundity, breeding strategy, age-at-maturity and mean trophic level. They also identified seven Susceptibility attributes; and overlap, migrations, aggregations, and other behaviour responses, morphological characteristics affecting capture, survival after capture and release and management strategy. Each species were ranked against low, medium and high Productivity and Susceptibility. They did not attempt to weigh the importance of each attribute, as is usually done in such assessments (e.g. Patrick *et al.*, 2010). They examined three management scenarios by varying the Susceptibility rankings but not the Productivity rankings, as these represent the intrinsic biological parameters of each individual species. They compared the Susceptibilities to these 16 fish species under the current management scenario, where all fishing below 1000 m depth is banned and thirdly where all fishing is banned during the blue ling (*Molva dypterygia*) spawning season. In comparison to the current management scenario, the fishery restrictions imposed in the other two scenarios reduced significantly the Susceptibilities to fishing to some shark species as these were found chiefly below 1000 m depth (scenario 2) and also to blue ling (scenario 3). For the remaining species, there were small or no changes in Susceptibilities between management scenarios.

Williams *et al*. (2011) applied the PSA methodology to evaluate the risk of fisheries to benthic habitats, and is conceptually similar to those developed at the time for fish species (e.g. Patrick *et al.*, 2010). They assessed the risk of a multisector fishery on 158 benthic habitats off southern Australia. The study area was very large with a depth range of 25 to 1300 m and encompassing a very large number of habitat types. The Susceptibility of the habitat was represented by three aspects, Availability, Encounterability and Selectivity. Each spectrum is in turn represented by a set of attributes that were ranked as reflecting low, medium and high risk (determined by the degree and the type of interaction with fishing). As an example, one of the attribute for the aspect Encounterability is ruggedness of the habitat that is categorized into three groups according to its relief. Productivity attributes, reflecting the intrinsic habitat properties, were represented by a single aspect. Using this approach, they provided risk estimation (high to low) to several habitat types, separated by fishing gears.

5.2.1.3 Level 3: Sustainability Assessment for Fishing Effects (SAFE)

Level 3 comprises a risk assessment according to the SAFE framework of Zhou *et al.* (2007). Level 3 is not a straightforward extension of level 2 methodology. The SAFE framework includes two components: indicators and reference points. One single indicator is used: fishing mortality rate. Limit reference points are established based on simple life-history parameters to avoid the obstacle of formal stock assessment that requires more extensive fishery and fishery-independent data. Thus, fishing mortality u is treated as a generalized property, not dependent on time since in most cases time-series data were pooled to derive proper abundance and distribution estimates, but on space (Zhou and Griffith, 2008). It considers the fraction P_N of the stock subject to a fisheries, which is characterized by catchability q and catch mortality 1-E:

$$u = P_N q(1 - E)$$

For rare species and data poor, P_N is based on likelihood estimates of abundances in fished and unfished areas, often from pooled surveys to create a sufficient database. To properly assess a stock, knowledge of life-history traits is required to determine the reference points MSY and the maximum sustainable fishing mortality a population can sustain before it will go extinct (u_{crash}).

Data requirements

Limited knowledge of the distribution of species in terms of survey data must be available. Life-history data and fisheries data must be known to an extent to model them properly (catchability, intrinsic growth rate).

Strengths/weaknesses

Sustainability aspects can be included on a population level. Trends in fisheries can be included in the analysis but trends in stocks are poorly discerned as either spatial characteristics are applied (Zhou *et al.*, 2007) or available time-series were pooled (Zhou and Griffith, 2008).

5.2.2 The MarLIN Approach

The Marine Life Information Network (MarLIN) has developed a methodology for the assessment of the sensitivity of several habitats/species populations to different environmental factors[1]. According to this approach, biotopes (i.e. habitats)/species population sensitivity depends on the intolerance of a species or habitat to damage from an external factor and the time taken for its subsequent recovery. Thus, two main features are assessed when estimating habitat/species population's sensitivity: 1) Intolerance which is the susceptibility of a habitat, community or species, to damage or death, from an external factor and 2) Recoverability: the ability of a habitat, community, or species to return to a state close to that which existed before the activity or event caused change. Since the response of a habitat/species population to a change in an environmental factor depends on the magnitude, extent and duration of that change, MarLIN developed a suite of standard levels of magnitude and duration of change (benchmarks) for 24 different environmental factors, against which the level of response of species and biotopes has been assessed. These environmental factors are those components of the physical, chemical, ecological or human environment that may be influenced by natural events or anthropogenic activity. In agreement with the benchmarks definitions, six "environmental factors" may be directly related to fishing impact, i.e. substratum loss, changes in suspended sediment, physical disturbance and abrasion, displacement, selective extraction of species.

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[1] For more detailed information, it is recommended to consider the full reports and peer-reviewed publications describing the methodology at the MarLIN web site (www.marlin.ac.uk).

Sensitivity assessment of species

Preceding the assessment, key information of the species is collected, with assurance of reliability of data. The sensitivity assessment of species comprises three major steps:

1) First of all the intolerance of species to an external factor arising from human activities or natural events is judged. The likely intolerance of the species is assessed with respect to a specified magnitude and duration of change for the aforementioned environmental factors related to fisheries impact. The assessment of intolerance is then made according to an intolerance scale[2] by reference to the reported change in environmental factors and their impact, relative to the magnitude and duration of the standard benchmarks and other relevant key information.

2) Next, the rationale assesses the likely recoverability of the species following cessation on the human activity or natural event. The likely recoverability of a species from disturbance or damage depends on its ability to regenerate, re-grow, recruit or recolonize, on the extent of damage incurred and hence its intolerance. The recoverability of species is assessed against the recoverability scale[3].

3) Finally, intolerance and recoverability are combined to provide a meaningful assessment of their overall sensitivity to environmental change. The overall sensitivity rank is derived from the combination of intolerance and recoverability using different scenarios according to a scale that is intuitively weighted towards recoverability. However, where recovery is likely to occur in a short period of time, intolerance has been given a greater weight rather than underestimate the potential sensitivity of marine species.[4]

Sensitivity assessment of habitats

The MarLIN approach to the assessment of the sensitivity of habitats assumes that the sensitivity of a community within a habitat is dependent upon and, therefore, is indicated by the sensitivity of the species within that community. The species that indicate the sensitivity of a habitat are identified as those species that significantly influence the ecology of that component community according to criteria that subdivide species into Key and Important based on the likely magnitude of the resultant change. The loss of one or more of these species would result in changes in the population(s) of associated species and their interactions. This initial part of the assessment applies to those species identified as indicative of habitat sensitivity. The first steps consist in a) the identification of the key species for each of the habitats and b) the preparation of a review of the biology and sensitivity key information for a habitat based in the aforementioned process for species sensitivity assessment.

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2 This scale can be found in the MarLIN rationale on the website. (http://www.marlin.ac.uk/sensitivityrationale.php)

3 This scale can be found in the MarLIN rationale on the website. (http://www.marlin.ac.uk/sensitivityrationale.php)

4 Definitions on sensitivity and the combination of Recoverability and Intolerance can be found in the Sensitivity rationale on the website. (http://www.marlin.ac.uk/sensitivityrationale.php)

After this first assessment of species Intolerance and Recoverability has been accomplished, the overall Intolerance/Recoverability can be derived from the Intolerance/Recoverability of the species identified as indicative of sensitivity, using the process described in the MarLIN Rationale[5]. Then the overall sensitivity rank is assessed for habitats from the combination of Intolerance and Recoverability. Finally the likely effect of the environmental factors on species richness is assessed. Indeed, change in an environmental factor may not significantly damage key or important species but may still degrade the integrity of the habitat due to loss of species richness. Therefore, the likely effect of the factor on species richness in the habitat should be assessed according to a ranking scale[6]. Where there is insufficient information to assess the Recoverability of a habitat the precautionary principle will be used and the recovery will be assumed to take a very long time, i.e. low recoverability in the derivation of a sensitivity rank.

Data requirements

The approach is quite data demanding and in many cases, the assessment of species/habitat sensitivity have been based on expert judgment.

Strengths/weaknesses

This approach is based on the assessment of species' intolerance in reference to benchmarks, and therefore cannot be used to take into account the effect of different gears or distinguish between pulse (i.e. single) or press (i.e. chronic) fishing disturbance. While not being focused on the effects of fishing on benthic habitats in terms of assessing multispecific interactions (e.g. effects on foodweb, etc.) the approach can be used to evaluate the sensitivity of habitats based on most important species' sensitivity. When integrated with information on fishing effort distribution it might be used to identify hot spots where fishing disturbance is likely to affect habitat sensitivity.

Case study

Tyler-Walters *et al.* (2009) applied and further developed the MarLIN Approach method to offshore sedimentary communities. The application of such method in this context is complex because of a lack of knowledge of the structural or functional role of many sedimentary species. This paper describes a method to assess the overall sensitivity of sedimentary communities, based on the intolerance and recoverability of component species to physical disturbance. A range of methods were applied to identify the best combinations of abundant, dominant or high biomass species for the assessment of sensitivity in the sedimentary communities examined. While being based on the same MarLIN approach for the species sensitivity assessment, this method adopts slightly different intolerance, recoverability and sensitivity classifications. Moreover, for the assessment at community level (i.e. habitat), the authors suggest to consider the five species that contribute the most to similarity and the ten species with the greatest abundance or biomass.

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[5] http://www.marlin.ac.uk/sensitivityrationale.php

[6] see MarLIN Ranking of response of species richness

5.2.3 US National Research Council approach

A US National Research Council report on the Effects of Trawling and Dredging on Seafloor Habitat (NRC, 2002) included a chapter on ecological risk assessment. This committee distinguished quantitative and comparative risk assessments. Quantitative risk assessment focuses on one risk at a time, and has been used in the policy arena to set standards and propose controls. The quantitative risk assessment framework was modified from an exposure-assessment model from the fields of human health and toxicology (NRC, 1993). This exposure assessment model has three phases: research, risk assessment and, risk management (Figure 5.2.3.1). The research phase includes, (i) process studies on the adverse effects of particular fishing gears on particular habitats, (ii) extrapolating process studies to areas that have not been studied, and (iii) field measurements of the spatial distribution of benthic animals and the frequency of bottom fishing. The risk-assessment phase includes (i) hazard identification, (ii) "dose-response" assessment, and (iii) exposure assessment. Risk characterization is the product of population density, mortality per tow and tow frequency. The risk-management phase places risk characterization in the appropriate regulatory framework and results in agency decisions and regulations.

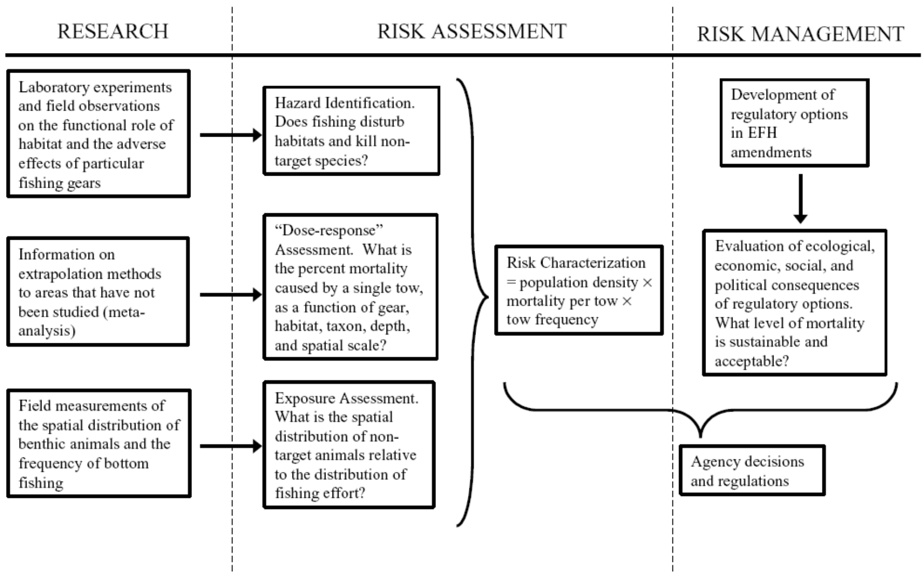

Figure 5.2.3.1. Elements of quantitative risk assessment (modified from National Research Council, 2002).

In comparative risk assessment, an array of ecological risks (e.g. different harvest scenarios) is identified by a group of stakeholders. After the risks are identified, they are compared with each other against a set of criteria chosen by the stakeholder group. These criteria might include the scale of the disturbance, level of scientific uncertainty, immediacy of threat, irreversibility, and species affected. A scale of highest to lowest is constructed using the criteria and relative risks are compared with these rankings. An example compared the risk to benthic fish habitat of fishery management alternatives in Alaska.

Space is implicit in these risk assessments given the spatial distributions of fishing pressure and species within habitats. Quantitative risk assessment would typically be applied one species at a time, but can be generalized to habitats.

Data requirements

An accurate risk assessment depends on fairly complete scientific information about the relationship between the stressor (e.g. trawling), the biologic community, and the suite of processes necessary to a functioning habitat. Comparative risk assessment can be applied when information is incomplete but scoring is inherently subjective and there is no consistent way to combine categorical scores to obtain an overall rank for each management option.

Strengths/weaknesses

Risk assessment is valuable for linking scientific knowledge to management and public values, which should be used to identify and prioritize risks. There is, however, no best method that should be applied to all ecological problems. The method chosen depends on the quality and quantity of scientific data available and the policy and social contexts of the problems to be addressed.

5.2.4 Extended overlap models

In this class of models a certain component of the criteria needed for a full risk assessment is lacking, and therefore may be regarded as 'reduced' risk assessments. However, they provide more information than a simple overlap analysis. As such, there are no straightforward measures of sustainability. In turn, these models provide pragmatic approaches to delineate categories of magnitude of impact through statistical properties of the distribution of resulting spatially resolved scores. In both cases described below a quartiles approach was chosen to define four impact classes. This gives managers the opportunity to balance impacts and mitigation in a spatially defined context and to define the present state.

5.2.4.1 Mortality and recovery data lacking

Conflicts between bottom-set gillnets and diving seabirds were analysed as the weighted overlap between fisheries and overwintering seabirds (Sonntag *et al.*, 2012). The conflict C was determined as the product of fisheries effort E and weighted bird abundance N in a given grid cell i:

$$C_i = V_i * E_i = \sum_k (N_{ik} * WF_{ik}) * E_i$$

The weight factor WF was determined according to Garthe and Hüppop (2004), using five factors related to species behaviour and status. It is noteworthy that this approach deals with a species ensemble incorporated into one index of vulnerability, most suitable also for benthic communities.

Data requirements

High quality knowledge of both fisheries and ecosystem components at the same resolution must be available. Good information on life-history traits is needed.

Strengths/weaknesses

Spatially explicit method for stocks where life-history parameters can only be provided in terms of ordinal categories to derive the weighting factor. No link to sustainability reference conditions possible, vulnerability scores provide only indirect evidence to infer strength of impact. Trends in fisheries and ecosystem component can be included (the latter only if monitored sufficiently). Available for a suite of ecosystem components inhabiting the same space.

5.2.4.2 Recruitment and distribution data lacking

The extension here is that substituting fine-scaled distribution data with model data based on a habitat model is applied and that fisheries is described in terms of actual catch. With no recruitment terms in their function, Stelzenmüller *et al.* (2011) analysed local vulnerability through the relationship:

$$V_i = p_i * \frac{Catch}{f_i}$$

Where p_i is proportion of stock in area unit I, and catch refers to the total catch in the reference area. Unity is a reference value with at $V>1$ local catch f_i being proportionally higher than the corresponding abundance, and at $V<1$ catch being lower proportional abundance. This approach links to the partial fishing mortality concept of Rijnsdorp *et al.* (2006).

Data requirements

Limited knowledge of distributional data required for ecosystem components. Spatially explicit and sufficient abiotic data to model habitat and fisheries are required.

Strengths/weaknesses

It provides spatially resolved model data for environments for which high resolution data are unavailable but sufficient habitat information is available. Experience with such habitat models is that they can be very unreliable, particularly at large-scales and therefore require extensive ground-truthing before they should be used in decision-making. Without analytical assessment, the model provides no link to sustainability reference conditions since mortality or recruitment processes are not accounted for.

5.2.5 Population level models

Risk-based assessments require explicit modelling of mortality and recruitment and of abundance data and their distributions, opposite to outlining the potential of risk in which scoring procedures is applied, e.g. in PSA (see Hobday *et al.*, 2007, p. 135). Zhou *et al.* (2007) and Fock (2011) provide assessment frameworks on the population level. The Zhou *et al.* (2007) is level 3 assessment in the framework by Hobday *et al.* (2007) and specifically designed to indicate risks in data poor stocks however with sufficient and extensive survey information. Only with a definite reference to the population-level assessments can address whether a fisheries/pressure is sustainable and thus be fully commensurable across a range of affected species or ecosystem components. With more detailed input, accuracy increases from score-based models to population level models (see 5.2.1).

Data requirements

Generally data hungry and most likely to be applied only in cases where sufficient survey data and biological information is available.

Strengths/weaknesses

Spatially explicit modelling of recruitment and mortality processes and link to sustainability reference conditions inherent.

5.2.6 Conclusions

There are many approaches to ecological risk assessment some of which have been detailed above. Often the approach taken depends upon data availability and/or quality. When data are very limited, it may not be possible to do more than an initial screening of the issues involving subject matter experts and stakeholders as appropriate. We have identified two strategies for undertaking this initial screening, i.e. the Hobday *et al.* approach (Section 5.2.1.1) and what we refer to as the "US National Research Council Approach" (Section 5.2.3). When there is some biological information and some knowledge of the habitat and fishery interaction then semi-quantitative assessments can be considered. The biological information must include key species and/or habitat so that information on their life-history traits can be assessed. The MarLIN approach (Section 5.2.2) and the PSA approach of Hobday *et al.* (Section 5.2.1.2) are two examples of what can be done with some information, although the "US National Research Council Approach" also provides some guidance in this type of situation. WGECO had the most reservations over these types of approaches because of the potential for poor information (selection and weighing of the input parameters) to be presented as having more reliability than it does, and thus running the risk of false outcomes to be produced. When there is high quality spatially resolved habitat and fishing effort data and good information on the life-history traits of the species involved, quantitative assessments can be made. These include extended overlap analyses (Section 5.2.4), population analyses (Section 5.2.5) as well as process studies on the adverse effects of particular fishing gears on particular habitats (US National Research Council Approach). In the view of WGECO only these approaches constitute an impact assessment in the sense that recovery and mortality rates can be fully modelled. For all situations it is critical that decision steps be fully documented to maintain transparency in order to increase confidence in the outcomes.

5.3 Background to NEAFC Request

NEAFC has identified sponge and cold-water corals to be indicator VME organisms for both new and existing fishing areas. Catch exceeding thresholds of 60 and 800 kg for corals and sponges prompts the move-on rule, which has been reviewed extensively by WGDEC (ICES, 2012). The choice of risk assessment for either existing or new fishing grounds would depend very much on data availability. In an ideal world, data on spatial distribution patterns of benthic habitat features and fishing effort on a high resolution and good understanding of the biology/life-history traits of the VME organisms would provide the necessary parameters to provide accurate spatially resolved risk assessment. Within the NEAFC Regulatory Area (RA), such high resolution data are generally lacking. Habitat mapping in the deep-sea environments is faced by logistical, practical and economic constraints. To date, relatively few areas of the NEAFC RA have obtained detailed information on the distribution and composition of VME species (ICES, 2012). These include parts of the Rockall and Hatton bank and Barents Sea (ICES, 2012). The overall footprint of the fishing effort within the NEAFC RA is reasonably well known. The footprint delineates the boundaries of the existing fishing areas. While some partner states do hold high resolution fishing effort data, the access of data could be limited. To evaluate the risk from fishing impacts on VME species, a good understanding on their life-history traits is required. This information is necessary to evaluate the risk of these VME organisms to be significantly adversely impacted by fishing activities. To date, there has been an extensive research on cold-water corals that can be used to determine their sensitivity to fishing impacts such as growth rates (Mortensen, 2001), and structural complexity (e.g. Söffker *et al.*, 2011). Knowledge of the reproductive potential and connectivity between populations is still poorly known (Miller *et al.*, 2010). Comparatively, the knowledge of the biology and taxonomy of sponges is much poorer. In the following section we provide some guidance of the choice of risk-assessment scheme for the NEAFC RA.

Key issues for the NEAFC fisheries assessment protocol

Priority should be given to fully quantitative assessments, and research should be allocated to increase knowledge levels sufficient to fulfil demands of such an analysis. Key elements are:

a) Habitat and species distribution patterns;
b) Distribution of fishing effort;
c) Assessment of sensitivity/vulnerability;
d) Uncertainty evaluation. There are a number of generic gaps in knowledge, such as:
  i) Pressure-state relationships;
  ii) Effects on deep-water habitats;
  iii) Classification of deep-water habitats still not complete;
  iv) Occurrence/distribution of habitats.

Since the points a), b) and c) will be outlined in detail in the following section, uncertainty analysis is elaborated on below.

Uncertainty evaluation

It is important that when evidence is used in decision-making, there is an explicit assessment of the quality of the evidence. This allows decision-makers to understand the risks of using that evidence. It is worth noting that evidence with lower certainty can still be used in decision-taking; insufficient certainty does not mean that decisions should be postponed. There have been a number of schemes for evaluating evidence; some of the most carefully reviewed are those deriving from the International Panel on Climate Change (IPCC 2005, 2010). One scheme to categorize evidence is shown in Table 5.3.1.

Table 5.3.1. Three categories of evidence certainty (from JNCC and Natural England 2011).

Low certainty
There is no direct evidence (peer-reviewed scientific, grey literature or non-scientific). It has been necessary to rely on analogy with other habitats for which evidence does exist. Evidence to support this assumption may be limited (i.e. the relative sensitivity of the habitats is not clear).
The feature may encompass a number of subtypes which vary in their sensitivity to fishing pressure. There is no direct evidence for any of the subtypes so it has been necessary to rely on analogy with several other habitats for which evidence does exist.
Conclusions have been based on sensitivity assessments which may rely on significant assumptions or generalisations. It has not been possible to validate these assumptions.
The evidence base is conflicting, as a result it is not possible to reach accurate conclusions on the effect of activities on features and consequently provide direct and clear advice.
Medium certainty
There is no direct evidence. It has been necessary to make an analogy with other habitats for which evidence exists. There is good reason to believe that the analogy is justified (e.g. occurrence of species with similar characteristics).
The feature may encompass a number of subtypes which vary in their sensitivity to fishing pressure. The available evidence does not cover the full range of the variation so some cases may not be well supported by evidence.
There is directly relevant scientific information to support the conclusion but it comes from grey literature sources.
There is relevant non-scientific information that directly support the conclusion on impacts and advice on management options.
High certainty
There is good quality, highly relevant scientific information to directly support the conclusion.
There is good quality, highly relevant non-scientific information that directly support the conclusion.
There may not be direct evidence to support the conclusions, but they are inevitable conclusions based on the application of common sense.

5.4 Response to NEAFC Request

Annex 5 of the Regulations of Bottom Fishing Activities in the NEAFC Regulatory Area states that assessments should address, *inter alia* seven elements. We have placed our suggestions for new elements that would be required to undertake an Ecological Risk Assessment, following one of the approaches detailed above, in association with the existing element with which they most closely align.

1) Type(s) of fishing conducted or contemplated, including vessels and gear types, fishing areas, target and potential bycatch species, fishing effort levels and duration of fishing (harvesting plan);

In addition to these elements the **timing of the fishing operations** should be provided. This is so impacts on spawning aggregations can be assessed as part of the risk assessment. It would be beneficial to also record the **location of the fishing activity** at the finest possible scale and ideally using VMS with pings at <2 hours (ICES, 2009a; Section 3 response to DGMARE special request, and see ICES, 2011b). This will facilitate fine-scale spatial management to enable both conservation objectives and fishing as has been initiated on Hatton Bank. Compilation and public dissemination of historical data on fishing locations would allow for a broader research community to undertake risk assessments.

2) Best available scientific and technical information on the current state of fishery resources and baseline information on the ecosystems, habitats and communities in the fishing area, against which future changes are to be compared;

Biological data at species and habitat levels can be partially derived from the BIOTIC database (Biological Traits Information Catalogue; www.marlin.ac.uk\biotic) by MarLIN (The Marine Life Network Information for Britain and Ireland). This database contains information on over 40 biological traits categories on selected benthic species, together with additional supporting information, including the bibliography of relevant literature from which the information was obtained. It provides an additional tool for data analyses in the field of benthic community ecology. For each species the general biology, taxonomy, distribution and habitat, reproduction and life-history traits are given.

These biological traits can be used in compliance with the characteristics and criteria identified by the FAO "International Guidelines for the Management of Deep-sea Fisheries in the High Seas" (FAO, 2009). These criteria aid identification of the vulnerability of species and habitats to bottom-contact fishing gears (FAO, 2009) (see Box 1). An example of the application of such criteria in the context of the NAFO Regulatory Area (NW Atlantic) can be found in Murillo *et al.* (2011). The authors scored against the above mentioned criteria some additional benthic invertebrate species, communities or habitat-forming species in the NRA, in order to evaluate whether they should be considered as VME indicators.

Box 1

i. Uniqueness or rarity – an area or ecosystem that is unique or that contains rare species whose loss could not be compensated for by similar areas or ecosystems. These include:
- habitats that contain endemic species;
- habitats of rare, threatened or endangered species that occur;
- only in discrete areas.

ii. Functional significance of the habitat – discrete areas or habitats that are necessary for the survival, function, spawning/reproduction or recovery of fish stocks, particular life-history stages (e.g. nursery grounds or rearing areas), or of rare, threatened or endangered marine species.

iii. Fragility – an ecosystem or faunal community that is highly susceptible to degradation by anthropogenic activities.

iv. Life-history traits of component species that make recovery difficult – ecosystems that are characterized by populations or assemblages of species with one or more of the following characteristics:
- slow growth rates;
- late age of maturity;
- low or unpredictable recruitment;
- high longevity.

v. Structural complexity – an ecosystem that is characterized by complex physical structures created by significant concentrations of biotic and abiotic features. In these ecosystems, ecological processes are usually highly dependent on these structured systems. Further, such ecosystems often have high diversity, which depends on the structuring organisms.

3) Identification, description and mapping of VMEs known or likely to occur in the fishing area;

Good data on distribution patterns and composition of the individual VME species is a prerequisite to evaluate their risk to fishing impacts. In almost all cases, data on the distribution patterns of the VME species in the NEAFC RA is scanty outside the existing fishing areas. Habitat modelling may be useful in making predictions of possible VME occurrences (e.g. Howell *et al.*, 2011). However, model predictions on spatial distribution patterns can be heavily influenced by the data availability and require extensive ground-truthing before they can be used in decision-making.

4) Identification, description and evaluation of the occurrence, scale and duration of likely impacts, including cumulative impacts of activities covered by the assessment on VMEs;

To examine the impact of fishing on the VME species/habitats a **high resolution data on fishing effort** is required. This would allow direct comparisons of the overlap between the distribution patterns of habitats and the fishing effort. Such data could also allow extended overlap analysis to be carried out. These approaches would further allow exploring various management scenarios, such as where the level of fishing effort is varied or the degree of overlap with the VME species occurs (Stelzenmüller *et al.*, 2011).

The identification of possible impacts (should an activity go ahead) is a critical part of any EIA process. This enables both a judgment of whether the impacts are acceptable, and whether any mitigation is required. It is difficult to describe whether an impact is "likely" as this depends on mitigation and other later decisions. However there is probably no need to describe possible impacts that are very unlikely to occur. It is recommended that the identification of impacts is not confined solely to VMEs – partly because not all VME habitats have yet been identified and partly that identification of the broad effects of activities is needed under the ecosystem approach to fisheries management.

The first step in this process is a description of possible fishing activity that may occur. The effects of these activities may then be described in terms of possible pressures caused by those activities (e.g. bottom trawling may cause pressures such as abrasion, removal of biota, removal of structural features, smothering) (Robinson and Knights, 2011) (See ToR f, this report). Impacts on habitats may then be described in terms of their responses to these pressures. It is important that possible activities are described in as much detail as possible; including in both time and space (e.g. trawling will potentially occur three times a year in a particular location). This is important as the intensity and frequency of an activity can affect the scale of the consequent pressure(s) and the resultant impact.

There are several approaches to evaluate the sensitivity of VME species. Tyler-Walters *et al.* (2009) propose an approach to evaluate sensitivity of benthic species in offshore sedimentary environments, which requires good knowledge of their biological traits, such as growth rates, generation times and age-at-maturity. Due to the much less knowledge of the functioning of benthic organisms and habitats in deep-sea habitats/species found in the NEAFC RA, this method may be unsuitable. Murillo *et al.* (2012) reviewed in total 500 taxa to evaluate whether they met the criteria for an indicator VME species. For this purpose, they used two approaches. Firstly, they ranked the species groups based on their biological traits as outlined in the FAO Guidelines (FAO, 2009). This approach considers attributes that reflect traits such as fragility, vulnerability and recoverability and evaluation on how significant role these have in the ecosystem. The other approach they used was to use indices of uniqueness and rarity.

5) Data and methods used to identify, describe and assess the impacts of the activity, the identification of gaps in knowledge, and an evaluation of uncertainties in the information presented in the assessment;

There have been a number of studies and reviews of the effects of fishing activity on habitats (e.g. Collie *et al.*, 2000; Eno *et al.*, 2001; Hinz *et al.*, 2009; Løkkeborg, 2005), but rather few of these have described impacts in deeper waters such as those that predominate in the NEAFC Regulatory Area; with the exception of effects of fishing on deep-water corals and other structural habitats. The overall effects of a pressure from fishing also depends on the intensity and persistence of that pressure; some habitats are more sensitive to pressure than others, and habitats recover (or return towards previous conditions) at differing rates following the occurrence of the pressure. The precise relationship between intensity of a pressure and the resulting state of a habitat is poorly known for most habitats. However, many deep-water habitats/benthic communities are highly structured and hence are likely to be more vulnerable to trawling impacts and demand longer recovery times (Hinz *et al*., 2009). Box 2 (FAO, 2008) indicates when an impact is assessed as significantly adverse. All of these issues create uncertainty in the data and evaluations of effects.

Box 2 (FAO, 2008)

The methodology for identification, description and evaluation of possible impacts (see bullet point 4 above) should evolve from the definition of Significant Adverse Impacts (SAI) as stipulated in the FAO Fisheries and Aquaculture Report No. 881 of 2008, which was produced on the basis of the UN Resolution 61/105. According to this report SAI are those that "compromise ecosystem the ability of affected populations to replace themselves; (ii) degrades the long-term natural productivity of habitats; or (iii) causes, on more than a temporary basis, significant loss of species richness, habitat or community types. Impacts should be evaluated individually, in combination and cumulatively". Moreover "when determining the scale and significance of an impact, the following six factors should be considered:

1) the intensity or severity of the impact at the specific site being affected;
2) the spatial extent of the impact relative to the availability of the habitat type affected;
3) the sensitivity/vulnerability of the ecosystem to the impact;
4) the ability of an ecosystem to recover from harm, and the rate of such recovery;
5) the extent to which ecosystem functions may be altered by the impact;
6) the timing and duration of the impact relative to the period in which a species needs the habitat during one or more of its life-history stages."

In 2008–2009 WGECO collaborated with WGFTFB to develop a methodology to assess and quantify the efficacy of Gear Based Technical Measures (GBTMs) to reduce environmental impact of fishing (ICES, 2008a,b; 2009a,b). Four ecosystem components were listed for assessment: (1) commercial fish species, (2) species previously listed as vulnerable or at risk, (3) marine mammals, reptiles and seabirds and (4) pelagic and benthic habitats. The framework details what constitutes a SAI and a non-SAI for those components according to available research and data. The developed case studies did not apply for the deep sea, but we recommend that the prescribed specifications and data requirements are reflected in methodologies used to evaluate possible impacts and knowledge gaps.

It is important that when evidence is used in decision-making, there is an explicit assessment of the quality of the evidence. This allows decision-makers to understand the risks of using that evidence; it is worth noting that evidence with lower certainty can still be used in decision-taking; insufficient certainty does not mean that decisions should be postponed. There have been a number of schema for evaluating evidence; some of the most carefully reviewed are those deriving from the International Panel on Climate Change (One scheme to categorize evidence is shown in Table 5.3.1).

6) Risk assessment of likely impacts by the fishing operations to determine which impacts on VMEs are likely to be significant adverse impacts;

There are a number of approaches which can be followed to risk assessment that can be followed:

Initial assessment

The initial assessment (Scale intensity consequence analysis (Hobday *et al.*, 2011) and the comparative risk assessment of the US NRC approach) could be used to explore various management scenarios in areas where there is poor data, as it is based entirely based on expertise judgment. As a practical example for the NEAFC RA, this could involve expert judgment of the effects of varying the fishing effort by otter trawl and longline on the VME organisms. The SICA analysis employs the "plausible worst case" approach to evaluation of risk, i.e. in the example above, the scenario with the highest score would involve the combination of the worst outcome of the exposure of the gear and the impact and recovery of the organism. This approach is feasible for a rapid evaluation even when there are large number of potential threats and impacts. As an example, Hobday *et al*. (2011) assessed 31 subfisheries that comprised a large number of fishing gear types.

Semi-quantitative assessment

The Productivity-Susceptibility Analysis (PSA) has been suggested to be a useful tool to identify habitats that are at risk to fishing activities (Williams *et al.*, 2011) although WGECO has reservations which are expressed in Section 3.2.1.2 above. This analysis could provide generic estimates on the susceptibilities of different habitat types or individual key VME species to fishing impacts. Further, the effects of different management scenarios could be explored, such as the effects of varying the level of the fishing effort or its spatial distribution. Due to the fact that the scores for individual attributes are additive and non-quantitative, their selection and weighing needs to be well supported. The risk of this approach is that the final scores for the susceptibility and productivity for a given habitat could be influenced by what attributes are included and how these are weighted. Evaluation of the quality of evidence is needed.

Extended overlap analysis and fully quantitative risk assessment

Both approaches require high quality spatially resolved habitat and fishing effort data, but differ in their need for biological data. These approaches quantitatively combine information on the distribution patterns of a VME species, the evaluation of their importance/sensitivity/functioning in the marine ecosystem and their threats. Such maps would be of immense importance when it comes to evaluate different management scenarios in the NEAFC RA and also to show the degree of spatially assigned uncertainty.

As data availability largely determines the options available for risk assessment, we provide Table 5.3.2 summarizing those options.

Table 5.3.2. Guidance on the usage of risk assessment methodologies based on data availability.

Data availability/quality	Type of assessment	Comment	Reliance on expert judgment
Data poor – no detailed distribution information	Initial screening	Assessments ending at this stage must be accompanied by documentation supporting the decision-making. Where risk is identified, one must move to a higher level of assessment.	High
Some biological information and some knowledge of interaction available	Semi-quantitative assessment	Requires information on biologically relevant components of the habitat and fishing effort and how these interact. Effort and habitat maps are highly desirable but not essential. Some evaluation of the overlap between the fishing and the habitat is required or it has no relevance.	Intermediate
High quality spatially resolved habitat and fishing effort data. Good biological information on life-history traits and knowledge of interaction between fishing and habitats available.	Spatially resolved assessment: extended overlap analysis	Habitat and fishing effort maps. The priority of this type of assessment is to evaluate the overlap in the spatial distribution of habitats and fishing effort. This analysis incorporates the traits and sensitivity of the habitat component. This analysis incorporates the traits and sensitivity of the habitat component. This should be quantitative.	Low
High quality spatially resolved habitat and fishing effort data. Quantitative information on life-history traits and quantitative knowledge of the interaction between fishing and habitats available.	Fully quantitative risk assessment	Habitat and fishing effort maps. Fully quantitative information on the biological traits and sensitivity. The priority of this fully quantitative assessment is to accurately evaluate the overlap between the fishing effort and the distribution of the habitats. Fully spatially resolved risk assessment is possible, as the recovery and mortality rate can be fully modelled.	None

7) The proposed mitigation and management measures to be used to prevent significant adverse impacts on VMEs, and the measures to be used to monitor effects of the fishing operations;

NEAFC would benefit if they request to review their assessments.

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6 ToRd) Indicators of Foodweb condition in reference to MSFD Descriptor 4

Examine and report on developing foodweb condition indicators, including those currently envisaged in the MSFD Communication.

Summary

As noted in the EC (2010) Decision Document on MSFD indicators, there are a number of shortcomings in the list of indicators proposed to support MSFD Descriptor 4. The principal perceived shortcomings were that the indicators primarily focused on measures of the "structure" of foodwebs, and only one indicator (on productivity) was proposed for the "functioning" or the dynamics of the processes operating within foodwebs. In this section, WGECO shows that these two characteristics of foodwebs are intertwined; variation in the structure of foodwebs invariably alters the dynamics of processes and therefore the functioning of foodwebs. Logically, the functioning of an ecosystem cannot directly be conserved, only the structures providing the functions can. Most structures will be subject to influences other than foodweb function. WGECO has developed two lines of thought: that GES of foodwebs is to a large extent dependent on foodweb function, and that GES of foodweb is largely reducible to questions of structure. Subsequent sections contrast and synthesize these views.

6.1 Introduction

In fulfilment of requirements of the Marine Strategy Framework Directive (MSFD; EC, 2008) the European Commission Joint Research Centre (JRC) and ICES formed expert Task Groups to provide advice on criteria and methodological standards to allow consistency in approach in evaluating the extent to which Good Environmental Status (GES) is being achieved. Task Group 4 addressed MSFD Descriptor 4 ("Foodwebs"), which reads

> *All elements of the marine foodwebs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity.* (EC, 2008).

The report by TG4 (Rogers *et al.*, 2010) is open about its limitations. They see Descriptor 4 as "one of the most difficult to implement", and highlight in many places ideas that are noteworthy and should be taken into account when considering revisions of the Criteria and Methodological standards in future. One of the practical decisions TG4 had to take was to narrow down the set of Attributes of foodwebs that they list (which correspond to the "Criteria" of EC, 2010) to those for which practicable Indicators could be found. TG4 emphasize the preliminary nature of their recommendations.

The list of indicators proposed by TG4 was further shortened in public consultations before their publication by EC (2010), and there remain sentiments (detailed in Section 6.2) that the list that entered the final document has its own specific shortcomings. EC (2010), too, are explicit that "Additional scientific and technical support is required, at this stage, for the further development of criteria and potentially useful indicators to address the relationships within the foodweb". The present report is a contribution to this development of the Descriptor, with the goal of supporting the revision of the Criteria and Methodological Standards scheduled for 2018.

To cope with the complex nature of foodwebs, WGECO set out to approach the problem of identifying indicators for foodweb condition in two steps. The first step is to identify and list attributes of foodwebs that deserve conservational attention (hereafter "attributes"). The population of this list with proposals for operational indicators comes only as a second step. It is likely that indicators proposed by EC (2010) can satisfy some of these roles, that for some attributes one or several other operational indicators can be identified, but that for other attributes no operational indicators are currently available, highlighting the need for research activity. The list of attributes itself is meant to be open to constructive criticism. Our two step approach enables a separation of the discussion on the relevant attributes of foodwebs from that on the appropriate indicators.

Deliberating the first step, WGECO encountered an issue that already shines through in the report by TG4. To what extent is the functioning of foodwebs an attribute that deserves conservation in the context of the MSFD, and to what extent is it just a diagnostic for attributes of foodweb structure? The issue and its implications are discussed in Section 6.3. In support of this discussion, Section 6.4 highlights possibilities for identifying the general relationships between foodweb structure, function, and dynamics by means of generic foodweb models, which may be a means of resolving the structure-function dichotomy. Section 6.5 develops suggestions by TG4 to determine indicators of foodweb function indirectly by combining data on state with foodweb models. In Section 6.6 we briefly review recent developments in work to identify abundances of functional groups as a means to characterize foodweb condition. In Section 6.7, a list of structural attributes of foodwebs is introduced, including an explanation for the underlying rationale. Conclusions and recommendations for ways forward at offered in Section 6.8.

6.2 Gaps and shortcomings of current MSFD foodweb indicators

The indicators selected by EC (2010) are based on three criteria related to the structure and energy transfer in foodwebs:

1) productivity (production per unit biomass) of key species or trophic groups (criterion 4.1);
2) proportion of selected species at the top of foodwebs (criterion 4.2); and
3) abundance/distribution of key trophic groups/species (criterion 4.3).

Whereas criterion 4.1 and the associated indicator 4.1.1., expressed as "predator performance" are proposed as measures of energy flow, structural properties of foodwebs are considered in criterion 4.2 and indicator 4.2.1, defined as "the proportion of large fish", and criterion 4.3 and indicator 4.3.1., expressed as "the abundance trends of functionally important selected groups/species".

By the understanding of TG4, Descriptor 4 aims to focus more on the functional rather than the structural attributes of the ecosystem as the other biodiversity descriptors (D1, D2 and D6); although structural indicators are essential to describe foodwebs, they may provide only partial information about trophic functioning. Yet, it is widely believed that in order to assess GES of Descriptor 4, other essential properties of foodwebs, especially other aspects of functioning, such as trophic efficiencies, and dynamics, such as resilience and stability, should be taken into account but are currently underrepresented or not covered by the proposed indicators of the Commission Decision (2010).

In relation to energy flow, for example, indicators that explicitly measure ecosystem processes (e.g. nutrient and material cycling, ecosystem metabolism) can provide complementary information to the currently proposed indicators (Bunn and Davies, 2000; Elliott and Quintino, 2007). Furthermore, indicators related to dynamic concepts of foodwebs, such as adaptation and resilience to perturbations, can be used to assess ecosystem health. Resilience can be defined as the magnitude of disturbance that can be tolerated before a system moves into a different region of state space and a different set of controls, as originally conceived by Holling (1973; 1996). Measures based upon the concept of resilience differ in two important ways from traditional indicators: they apply to the entire system (its functioning and organization) and they focus on variables that underlie the capacity of the system to provide ecosystem services, whereas other indicators often address only the current state of the system or service (Carpenter *et al.*, 2001). However, while these statistical indicators can be useful tools for empirical studies of non-linear dynamics of ecosystems or other complex systems, operational measures for ecosystem management are needed (e.g. Rombouts *et al.*, under review).

Current criteria and indicators for D4 focus on single components of the foodweb (e.g. key predator) or a proportion of species at the top of the foodweb, and are unlikely to capture whole system energy flow. For example, indicator 4.1.1 assumes that the performance of key predators reflect transfer efficiencies of underlying trophic levels and thus can be used to infer the integrity of the foodwebs. However, in reality, the dynamics of key predators are often influenced by other intrinsic (e.g. predation and consumption rate, feeding habits, etc.) and extrinsic factors (e.g. climate variability, availability of breeding sites, disease) than food availability alone. If top predators are not food limited, then variations in their abundance or performance infers nothing about the underlying processes occurring at lower trophic levels. Hence, the isolated use of population indicators is likely inappropriate to define GES of foodwebs.

The most advanced developments of foodweb indicators stem from a top–down view, e.g. the perspective of fisheries management practices. However, other stressors than fishing can impact foodwebs, such as pollution (Boon *et al*., 2002), eutrophication (Cloern, 2001), habitat destruction (Melian and Bascompte, 2002) and climate change (Muren *et al*., 2005; Kirby and Beaugrand, 2009), and these can influence the foodweb differently (Moloney *et al*., 2010). Whereas the removal of top-predators due to exploitation will exert a top–down effect on the foodweb, excessive inputs of nutrients, caused by localized chemical and physical forcing, will mainly propagate from lower trophic levels up the foodweb and so have a bottom–up effect. Organisms located at lower trophic levels can be particularly useful as indicators of trophic functioning due to their important role in the production of organic matter (Livingston *et al*., 1997) and the transfer of this energy by primary consumers to other components of the foodweb (Beaugrand *et al*., 2008). Furthermore, as suggested by TG4, indicators that consider attributes of micro-organisms, benthic invertebrates, and that link these groups to higher trophic levels, i.e. benthic-pelagic coupling, are needed to obtain a more comprehensive image on the functioning of foodwebs.

The Commission Decision states that "the assessment and methodologies required need to take into account and, where appropriate, be based on those applicable under existing Community legislation". Hence, the choice of foodweb indicators from the TG4 review has likely been influenced by the availability of data to calculate these indicators. However, the objectives of many existing biological long-term monitoring programmes have been focused on providing biological information to allow the sustainable use of resources. As a result, many ecologically important species, groups and habitats are currently underrepresented but might nevertheless prove important to assess GES of foodwebs.

Despite an increasing importance of foodweb issues in marine conservation management, few tools and methods are currently in use to derive foodweb indicators (Rogers *et al*., 2010). Progress has been inhibited by the difficulty in measuring foodweb function and dynamics, such as fluxes, and the lack of knowledge of how foodweb structure relates to function and dynamics. In addition, the use of foodweb models for assessing foodweb indicators is constrained by the data requirements for specifying parameters, often large in number, and limited knowledge of the precise form of ecosystem interactions.

6.3 Function, structure, and dynamics

Logically, the functioning of an ecosystem cannot directly be conserved. Only the structures providing function can. TG4 recognize this relationship by explaining in their rationale for selecting "energy flows in foodwebs" as their Attribute 1 that "Energy flows through the foodweb are an attribute which allows us to diagnose the state of the system. "We value the foodweb function of transporting energy and carbon from lower to higher tropic levels, among others, because this supports the production and harvesting of fish. Future generations might instead put higher value on carbon sequestration, and so on the top–down control on zooplankton, e.g. by fish, which limits the transport of carbon to higher trophic levels. Assuming some form of resilience of foodwebs (e.g. Section 6.5 below), functioning will be impeded only after some period of structural degradation. Loss of function is the last step in a causal chain, and perhaps the one most difficult to attribute to particular human impacts.

Use of function (e.g. energy flow) as a diagnostic for the state of the system can be appropriate when

1) There is uncertainty towards what the structure supporting a function is; or
2) Changes in function are detected more easily than changes in structure.

In both cases, it is necessary to define a function directly, rather than inferring changes in function from structural changes (e.g. abundances of indicator species).

Function as an indicator for GES can be useful also when

3) Continued lack of function (e.g. removal of pollutants by biological organism) can have long-term effects on structure.

While points 1–3 are likely to play a role, the main reason why TG4 find that

> [Descriptor 4] *addresses the functional aspects of marine foodwebs, especially the rates of energy transfer within the system and levels of productivity in key components.*

seems to be different. TG4 argue that

> *Foodwebs are networks of feeding interactions between marine organisms. The species composition of foodwebs varies according to habitat and region, but the principles of energy transfer from sunlight and plants through successive trophic levels are the same.*

In this context it is worth remarking that, while the energy flow is recognized as an important aspect of foodwebs, it plays only a secondary role in foodweb theory. Instead, the theory is mired by issues of stability and coexistence.

A typical example is the question of what determines the length of food chains, related to survival of top predators. As May (1999) explains:

> *Why are food chain lengths so relatively constant and relatively short? The conventional explanation used to be that inevitable inefficiencies in energy transfer from one level to the next precluded the possibility of long-chains in which predator was piled upon predator. This explanation would suggest that food chains should be longer in highly productive environments with a larger energy base, and in communities of cold- rather than warm-blooded species (because the efficiency of energy-transfer between trophic levels is significantly higher for ectotherms than for endotherms). Neither tendency is observed. Recent work has explored a variety of other possibilities, including that food chain lengths may be controlled mainly by dynamical considerations, with long-chains leading to excessive levels of population fluctuation. As I see it, this is a central question for ecological science.*

Perhaps then, we consider the foodweb function of transferring energy from lower to higher trophic levels to be important, simply because of ecosystem services this function provides to society?

WGECO took the pragmatic approach to develop both lines of thought: That GES of foodwebs is to a large extent dependent on foodweb function, and that GES of foodweb is largely reducible to questions of structure. Subsequent sections contrast and synthesize these views (see Sections 6.4 to 6.7).

Over and above this, a third recurring topic of foodweb science should not be forgotten: the dynamic properties of foodwebs at the community level, in particular issues of stability and resilience (see Sections 6.2, 6.5).

6.4 Linking function, structure and dynamics through models

As identified in Section 6.2, the existing MSFD foodweb indicators focus on measuring structural rather than functional or dynamical aspects of marine ecosystems. However, *a priori*, it is unclear whether structure and function provide equivalent information. As a means of resolving the structure-function dichotomy, models can be used to understand the general relationships between structure and function. The use of models also helps to understand the dynamical aspects of foodwebs. This section first describes how mathematical process-based models can be used to examine how ecosystem structure relates to function and dynamics under a variety of pressures. Two multispecies size-structured models, identified in the last WGECO report (ICES, 2011), are then examined in more detail, with provision of worked examples of their use in assessing the relationship between specific structural, functional and dynamical indicators.

6.4.1 The heuristic value of models

Controlled experiments have been used to quantify the relationship between foodweb structure, measured as biodiversity, and ecological function and stability (Hooper *et al.*, 2005). However, such experiments have focused on terrestrial and freshwater ecosystems, and are typically confined to small spatial scales and one trophic level or taxonomic grouping (Hooper *et al.*, 2005). For large, complex marine ecosystems, whole foodweb experiments would likely be both unethical and practically unfeasible

(Rafaelli, 2006). On the other hand, in-silico experiments using numerical models of these marine ecosystems can be used to corroborate scientific hypotheses and conduct sensitivity analyses that examine "what-if" scenarios.

In order to assess linkages between structure, function and dynamics, models need to explicitly represent salient features of ecosystem structure and processes that allow ecological function and dynamics to emerge. A number of models with these features have been developed for an ecosystem approach to fisheries, reviewed by Plagányi (2007). Ideally, models would be developed that incorporate all elements of the complexity of marine ecosystems, and for specific regions around the world. However, this is not a practical possibility because of data requirements for parameterization and mathematical intractability. In addition, a site-specific modelling approach could mean that general ecological principles operating across sites are overlooked. It is not possible to simultaneously maximize the generality, realism and precision of models (Levins, 1966). Thus, use of different models, each giving a different perspective, is desirable. Results across models can be compared and contrasted, allowing effects of differences in model structure on conclusions to be determined, as well as synthesized.

In the context of foodweb indicators, some models can be particularly useful for:

a) Quantifying the relationship between structural indicators and indicators measuring ecological function or dynamics, thus allowing structural proxies of function to be identified (if any). To represent changes in structure, the models should not *a priori* fix the relationships.

b) Identifying indicators of function or dynamics that are sensitive and specific to scenarios of anthropogenic pressures, thus informing management of data requirements for monitoring of these indicators.

c) Setting thresholds and limits for indicators of function or dynamics (or their structural proxies) beyond which GES is not maintained.

The next subsection considers specific examples of how (a) and (b) can be achieved.

6.4.2 Specific modelling examples

Firstly, the Population-Dynamical Matching Model (PDMM) is used to demonstrate how models can be used to quantify the relationship between ecosystem structure and function. The PDMM is a dynamic species-resolved model that, uniquely, is able to generate foodwebs with hundreds to thousands of coexisting species, as found in real marine ecosystems. This is achieved by constructing foodwebs through a stepwise assembly algorithm that mimics the process of gradual community assembly in nature. The PDMM has been parameterized to represent a temperate shelf ecosystem in the Northeast Atlantic, allowing construction of model shelf communities with properties that match empirical data well (Shephard *et al.*, 2012). Importantly, model shelf communities have a realistic range of body masses, fish species richness, and average number of trophic links for fish species. This indicates that the model is able to capture real marine foodweb structure well. Further model details, including precise mathematics, are given in Rossberg *et al.* (2008) and last year's report by WGECO (ICES, 2011).

The PDMM permits construction of model communities with high fish species richness, as found in natural communities. In contrast, other model communities mostly have fewer than 30 species (Plagányi, 2007). Therefore, the PDMM is better suited for examination of how ecological function and dynamics change with the structural indicator of species richness, defined as the number of fish species in a community (Farnsworth *et al.*, 2012). Moreover, the PDMM can be used to examine the effects of loss of fish species on ecological function, which can represent stock collapses or local extinctions due to fishing (Dulvy *et al.*, 2003). Specifically, using a population of PDMM shelf communities, the variation of total fish production with fish species richness is quantified by sequential deletion of fish species. The ecological function considered, total production, is one that has been widely studied for terrestrial systems (Hooper *et al.*, 2005); in addition, total fish production has been used in a recent modelling study to measure part of the fish biomass available to fisheries (Garcia *et al.*, 2012). Changes in the sequence of species loss can lead to changes in the structure-function relationship (Solan *et al.*, 2004). Therefore, two different scenarios are examined:

i) random deletions; and

ii) deletions in order of decreasing body mass, representing "fishing down the foodweb" (Pauly *et al.*, 1998).

After each species deletion, the model system was allowed to dynamically relax to a new equilibrium, allowing full manifestation of indirect effects. Model results show that under both scenarios, initially, production does not change much with species richness (Figure 6.4.2.1). With deletions in order of decreasing body size (ordered species deletions), total fish production only starts to show a decreasing trend when the model fish community has lost about 50% of its initial richness (Figure 6.4.2.1). This suggests that for this fishing scenario, the functional indicator is only responsive to the pressure when fish species richness is at relatively low levels.

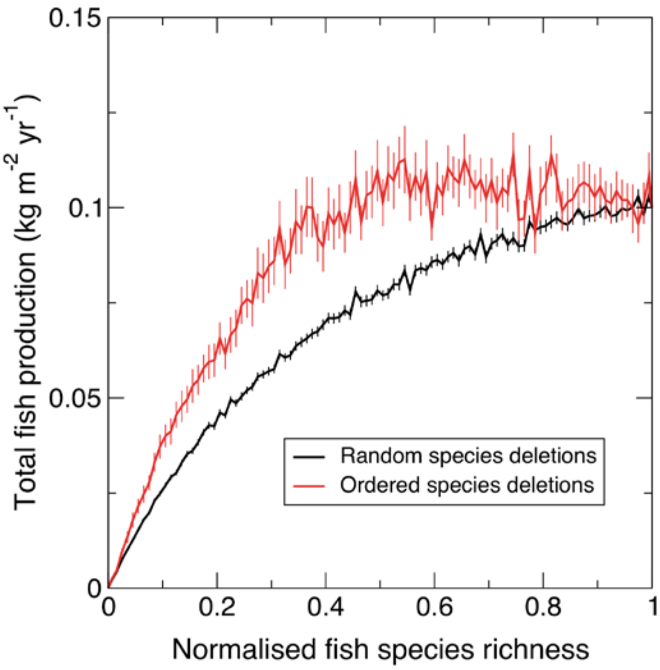

Figure 6.4.2.1. Relationships between total fish production, a functional indicator, and (normalized) fish species richness, a structural indicator. These are derived from species deletion experiments using a population of ten large model shelf communities, constructed using the PDMM. The model communities have 152–280 fish species. For each model community, ten random deletion experiments, where fish species are randomly chosen and deleted sequentially and one ordered deletion experiment, where fish species are deleted sequentially in order of decreasing size, were performed. The lines represent means and standard errors for all random deletion experiments (black lines) and all ordered deletion experiments (red lines).

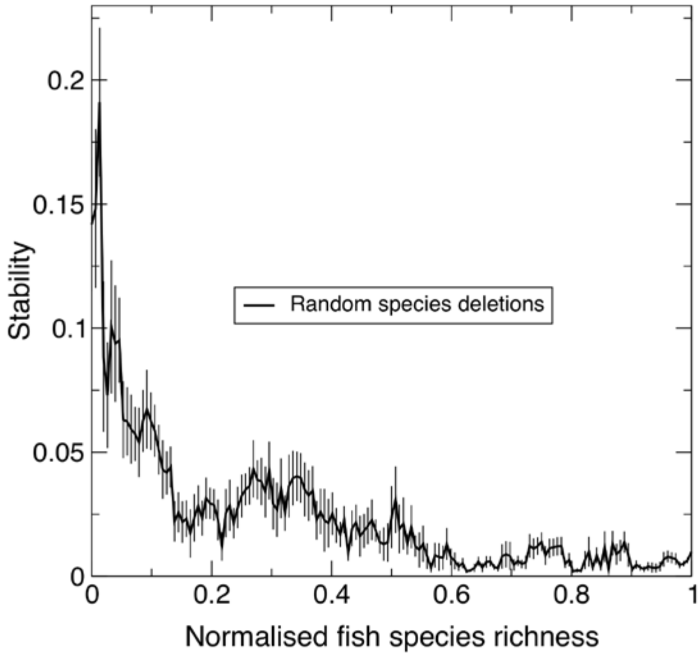

Figure 6.4.2.2. Relationship between stability and fish species richness, a structural indicator. Stability is measured as the return rate to equilibrium, as specified by the absolute value of the real part of the dominant eigenvalue (Robinson, 2004). The relationship is derived from ten random species deletion experiments using one large model shelf community, constructed using the PDMM. This model community has 152 fish species. The lines represent means and standard errors for all random deletion experiments.

Secondly, the PDMM is used to demonstrate how models can be used to quantify the relationship between ecosystem structure and dynamics. Indicators of ecosystem dynamics encompass indicators of ecosystem stability, which include the amplitude of fluctuations, the number of alternative stable states and the rate of return to equilibrium following a pulse perturbation (Ives and Carpenter, 2007). There is also a growing body of theoretical and empirical work that have identified early warning signals in dynamics leading up to a critical stress threshold or "tipping point", where an ecosystem becomes attracted to an alternative stable state (Scheffer *et al.*, 2009; Carpenter *et al.*, 2011). Such early warning signals include increased variance and skewness of dynamics; most relevant in a fisheries context are studies demonstrating increased variance of fish abundances due to fishing (Hsieh *et al.*, 2006; Anderson *et al.*, 2008). Preliminary analysis using one PDMM shelf community suggests that the return rate to equilibrium, following a localized perturbation, shows an increasing trend with decreasing fish species richness (Figure 6.4.2.2). This result is consistent with the classical theoretical work of May (1972), which used far simpler models with random interactions. The model result suggests that the structural indicator "fish species richness" might be used as a proxy for the stability indicator "return rate to equilibrium". Testing this hypothetical relationship with empirical data analyses would increase its reliability for management guidance.

Thirdly, the Fish Community Size-Resolved Model (FCSRM; Hartvig *et al.*, 2011; Houle *et al.*, 2012) is used to demonstrate how models can be used to measure resistance to change, a component of resilience (Levin and Lubchenco, 2008). The FCSRM is a multispecies size-structured model that, in contrast to the PDMM, explicitly resolves the intraspecific size spectrum of each fish species, with the computational and mathematical trade-off being representation of fewer fish species. It is based on the model of Hartvig *et al*. (2011) and has been used to examine the sensitivity and specificity of a suite of fisheries indicators (Houle *et al*., 2012). Further details of the FCSRM are given in Houle *et al.* (2012). The FCSRM is used to examine resistance of trophic cascades (Frank *et al*., 2005; Daskalov *et al*., 2007). Under a scenario of equal fishing mortality on all fish species, a large trophic cascade occurs due to differential responses among species (Figure 6.4.2.3). Interestingly, the cascade remains nearly unchanged after a doubling of fishing mortalities on medium-sized and large fish species, indicating strong resistance to this additional pressure. However, if mortalities are doubled again, the trophic cascade dissipates, apparently because of population collapse of the largest fish species (Figure 6.4.2.3).

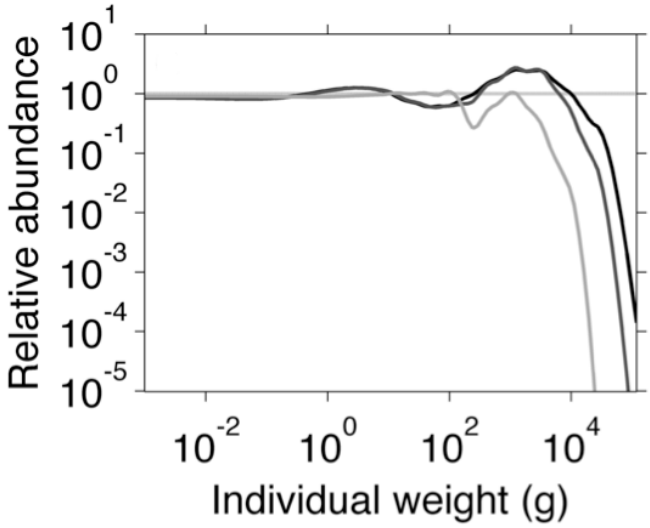

Figure 6.4.2.3. Effects of fishing on the community size spectrum, for a model community constructed using the FCSRM, with nine fish species. Three of these are small forage fish species (asymptotic body mass <500 g), three are medium-sized (asymptotic body mass between 500 g and 10 kg) and three are large (asymptotic body mass >10 kg). The baseline case has no fishing, against which relative changes in abundance are measured for three different fishing scenarios, with fishing mortality of 0.5 yr^{-1} on the three small forage fish species and fishing mortalities of 0.5 yr^{-1} (black line), 1 yr^{-1} (grey line) and 2 yr^{-1} (light grey line) on the other six fish species.

6.5 Fluxes as foodweb functions

To decide on attributes to be addressed by foodweb indicators, one that seems obvious to consider is the energy flux through the foodweb. To identify proper foodweb indicators of this kind it is important first to identify the foodweb concerned, by establishing information on its most important compartments, species and fluxes. For later use of the indicators in management, it is also important to include the major pressures or drivers of change. As recognized already by TG4, models can be used as helpful tools to estimate fluxes.

The key components are mostly generic and usually described by the trophic levels (e.g. plankton, benthos, fish, seabirds and sea mammals) or as functional groups (e.g. primary producers, secondary producers, intermediate consumers, top-predators and decomposers). The latter set of components is more generic. Using functional groups allows for indicators based on functional traits of organisms or life stages rather than the taxonomy.

As TG4 already noted, the components chosen as representative for the foodweb models needs to be considered very closely, as generic groups may be too wide and at the same time, they cannot easily be replaced with representative species. Well defined generic groups, such as functional groups, e.g. primary producers, secondary producers, intermediate feeders, top-predators, filter-feeders and decomposers, to optimize the predictive power of the model (see examples of models below).

The pressures are changes in environmental abiotic and biotic parameters (bottom–up drivers) and pelagic and demersal fishing or other human activities (top–down drivers). The bottom–up effects are nature driven and crucial to modelling energy flow (see Section 6.7). Having established the components and the pressures, one can establish the status of the different groups and the relationship or fluxes between them.

The relation between function and structure has previously been analysed by several authors. For example Heath (2005) estimated changes in the total secondary production demanded by fish between the 1970s and the 1990s, and finds strong indications for bottom–up control of the North Sea pelagic foodweb. Production of the benthos was more top–down controlled, with benthic invertebrates released from predation pressure as a result of fishing.

Building on these results Kenny *et al*. (2009) analysed statistical patterns relating ecological compartments such as plankton, pelagic and demersal fish stocks, and birds. Benthos, being an important compartment as well, was excluded due to lack of sufficient dataseries. A comparison between two decades (1983–1993, 1993–2003) showed a clear difference, with overexploitation present as main drive force (top–down) in the first decade and more environmental pressure (e.g. climate change) and relatively less fishing pressure in the second decade (bottom–up).

For foodwebs changes in fluxes are important, e.g. total primary production and the flux from this production to zooplankton, benthos, fish, birds, mammals, recycled trough spawning products, decomposers and as well as inorganic nutrients and trace minerals (Figure 6.5.1). However, those fluxes are difficult to measure in the field. Primary production data may be available from labelled CO_2 uptake or oxygen measurements, but the flux towards the higher tropic levels is more difficult to measure. New techniques like isotope distribution or fatty acid analyses give information on the pathways but they might be too time consuming or not differentiating enough to be used as an indicator of change. Therefore it is suggested to use model results to calculate foodweb parameters like primary and secondary production and the flow of energy or carbon between the different trophic levels and functional groups. For the North Sea, Ecopath, Ecosim and ERSEM models are available and could be used to test their applicability of this method. A recent noteworthy development is a new generation of DEB (Dynamic Energy Budgets) models (Kooijman, 2009). There are needs to further develop of applications of models for the purpose of estimation of fluxes in foodwebs.

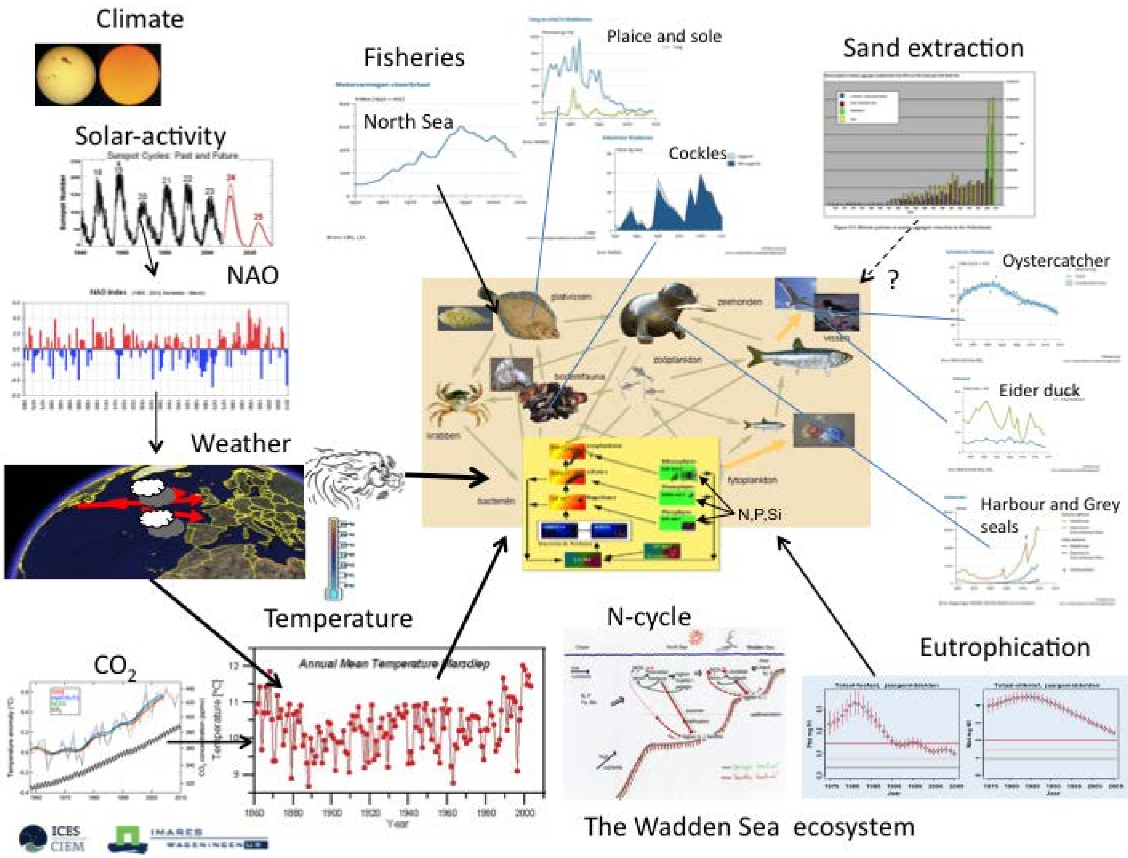

Figure 6.5.1. An ecosystem overview of the Wadden Sea, with pressures and long-term changes in different species. Presented by Han Lindeboom (unpublished).

6.6 Determining abundances of functional groups from survey data

The Marine Strategy Framework Directive requires indicators of "Abundance trends of functionally important selected groups/species (4.3.1)" to identify Good Environmental Status under Descriptor 4. Intensive exploitation of the target species usually at the highest trophic level can result in significant changes in the abundance and composition of the lower trophic levels, and thus the ecosystem as a whole. For example, major changes in the community structure of the Northeast Atlantic foodwebs have occurred (Pauly *et al.*, 1998). Consequently, the structure and composition of all trophic levels of the foodweb need to be considered, and not just those parts deemed to be commercially valuable. Assigning fish species within a foodweb to their respective feeding or trophic guilds, and monitoring any changes in the composition of these guilds over time, will incorporate all elements of the fish community.

Here, an example for making this approach operational is given (Paula Haynes, pers. comm.). In the North Sea, Greenstreet *et al*. (1997) assigned fish species to one of four broad trophic guilds (demersal piscivore, demersal benthivore, pelagic piscivore, and pelagic planktivore) based on adult diet data. However, fish can change diet as they grow in size, and this change is generally not considered in many trophodynamic studies. Demersal piscivores such as turbot (*Psetta maxima*) will begin life as a planktivore, before becoming benthivorous, and finally piscivorous, thereby occupying three trophic guilds throughout ontogeny. These shifts in the diet of species at different sizes need to be considered, in order to correctly allocate a fish to its appropriate trophic guild.

A Trophic Guilds look-up table for all Northeast Atlantic fish species, of every possible length was established (see Appendix 1). Using all available dietary data, fish were assigned to their respective trophic guilds, taking size into account. To test the applicability of this Trophic Guilds look-up table, trophic guild time-series were calculated for three fisheries surveys operating in the MSFD 'Celtic Seas' subregion, in order to describe changes in the broad trophic structure of the fish community as expressed in these survey data. The surveys included the UK West Coast Groundfish Survey, the Irish Groundfish Survey (Celtic Sea), and the Irish Sea Survey. To calculate changes in trophic structure, biomass data from each survey were combined using the Trophic Guilds look-up table. Results for the Celtic Sea area are presented below (Figure 6.6.1).

A previous study on the Celtic Sea computed another MSFD Descriptor 4 indicator, the Large Fish Indicator (LFI), which characterizes the size composition of the fish community, and found an overall reduction in the proportion of large fish in the area (Shephard *et al*., 2011). Interestingly, as demonstrated by the Trophic Guilds indicator used here, the change in size structure was not reflected by a change in the trophic guild composition. The biomass abundances of trophic guilds fluctuated without trends, as opposed the continuous decline in the abundance of fish found using the LFI. Therefore, the Trophic Guilds Indicator is providing complementary information to the LFI on the composition of the fish community. This approach, accounting for ontogenetic changes which occur in the diet of fish when assigning them to trophic guilds, does therefore allow for the identification of trends in the size composition of community structure over time.

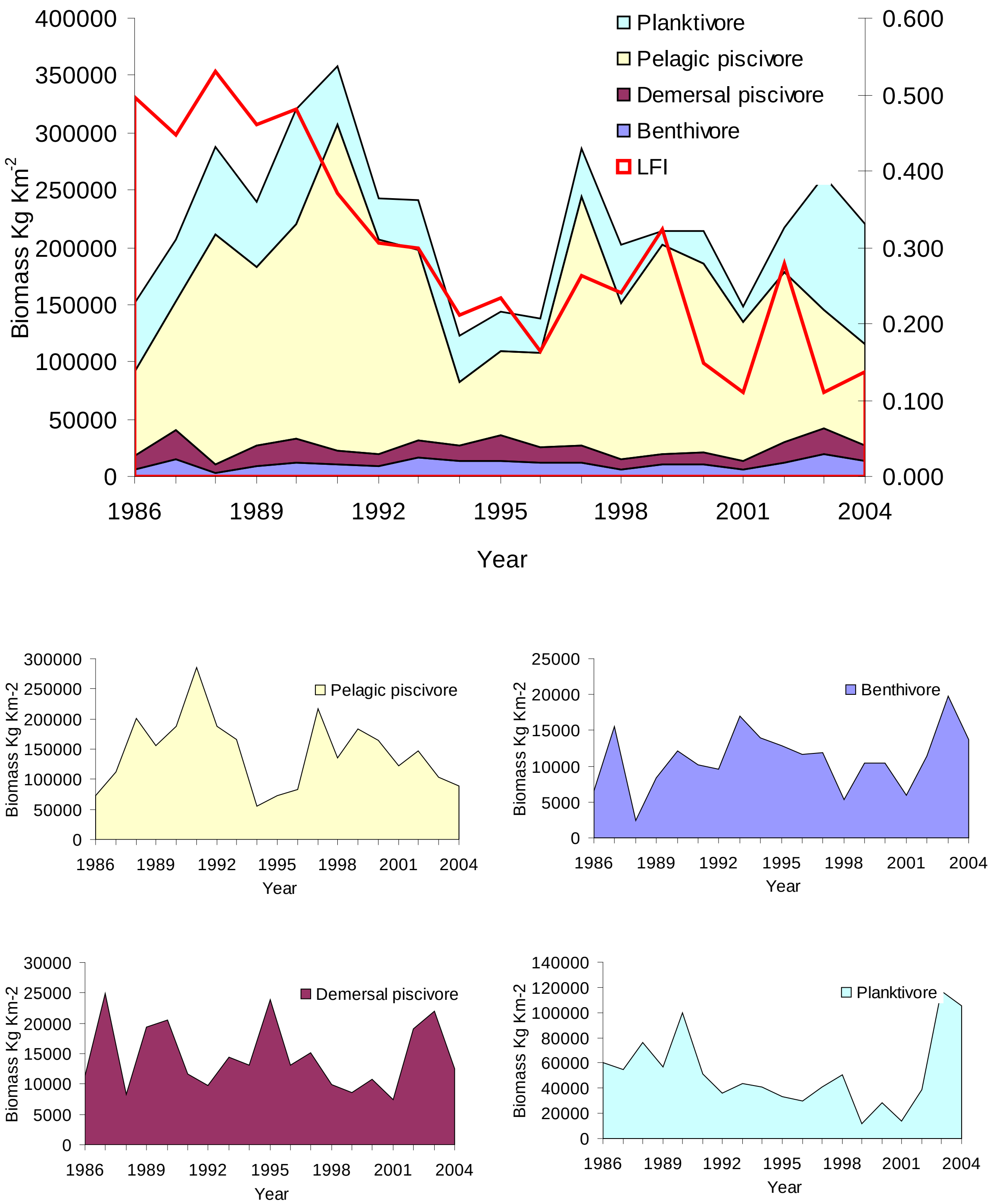

Figure 6.6.1. Trends in abundance (Kg Km^{-2}) of key functional groups of fish in the Celtic Sea fish community (WCGFS data). The LFI time-series is taken from Shephard *et al.* (2011).

6.7 Attributes of foodweb structure

The goal of this section is to develop a systematic approach towards selecting a set of attributes of GES for foodwebs for which, in a second step, appropriate operational indicators can be sought.

6.7.1 Rationale

Conservation is important for those aspects of ecological status for which natural recovery from pressures is either slow or impossible. For foodwebs, the question of what these aspects are has no obvious answer. Foodwebs are complex, the relevant spatial and temporal scales are often too large, and data too limited to base the answer on past observations alone.

It is characteristic for complex systems in general that some of their properties can be captured by models of some kind, while others cannot. For properties of complex systems that can be modelled, their responses to different scenarios of pressures can be predicted by models, guiding management choices. Properties whose responses cannot be predicted are more difficult to manage.

An approach to identifying relevant attributes of foodweb condition is to focus on those properties that can be described by models, and to select among them those for which recovery from pressures is slow or impossible. Fortunately, properties of complex systems with slow dynamics or hysteresis[7] (e.g. bi-stability) tend to belong to those accessible by models.[8]

6.7.2 Types of models

Natural foodwebs in their entirety are difficult to model. This simply reflects the fact that many aspects of foodweb structure can vary comparatively rapidly and unpredictably over space and time. Aspects of foodweb dynamics for which models could successfully be built appear to be restricted to the following three types:

i) models for community size structure;

ii) models reproducing statistical and dynamical patterns in the fine structure of foodwebs consisting of nodes (species) and links (feeding interactions);

iii) models for interactions between functional groups.

6.7.3 Attributes of GES: size spectra

Early dynamic models of community size structure in aquatic communities have been proposed by Benoit and Rochet (2004). Applications of such models have since proliferated (e.g. Hartvig *et al*., 2011; Rochet and Benoit, 2012; Houle *et al*., 2012). The predominant response to pressures found in these models is the classical downward trophic cascades (Figure 6.4.2.3). However, these go along with two other types of independent responses (Figure 6.7.3.1): Upward trophic cascades (a cascade formed by species larger than those under pressure; Andersen and Pedersen, 2010), and decline of the biomass of species larger than those under pressure, which intensifies along the size axis (Rossberg, 2012). Interestingly, only the last type of response necessarily leads to disruption of the energy flow from smaller to larger species. It is particularly important for perturbations at the bottom of the food chain (Section 6.5). Trophic cascades just re-channel the fluxes (Rossberg, 2012). The intrinsic time-scale

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[7] A system is said to exhibit hysteresis if its equilibrium state does not only depend on the current pressures but also on previous history.

[8] Hence, the idea is not that such properties are important just because there happen to be models, but that models could be built because the properties are important.

of the two upward effects is about ten times longer than that of the downward cascade (e.g. decades rather than years; absolute time-scales depend on the exploited size class).

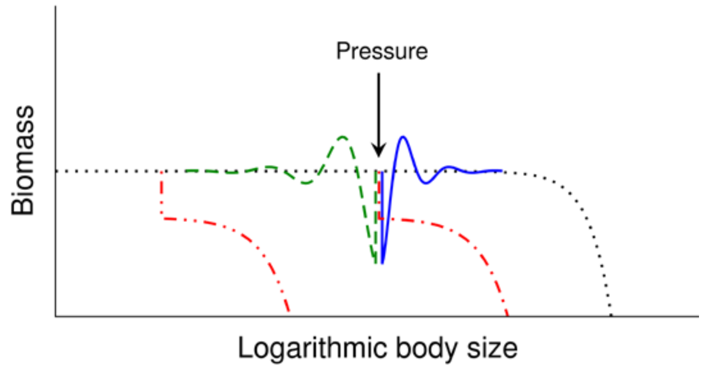

Figure 6.7.3.1. Schematic illustration of the three main responses of community size spectra (black dotted line) to size-specific pressures (arrow): the classical downward trophic cascade (green dashed line), an upward trophic cascade (blue solid line), and a depletion of biomass for all species larger than those under pressure, with increasing effect for larger sizes (right red dash-dotted line). The last response involves the disruption of energy flow from small to large species. Its impact can be particularly strong for pressures on small size classes (left red dash-dotted line).

Although the time-scales of downward cascades can be relatively short in theory, observations suggest that nevertheless recovery can be slow (Frank *et al*., 2005; 2011). The amplitudes of all three responses of size spectra to pressures therefore appear to be relevant attributes. However, it is currently unclear whether they can be separated empirically.

6.7.4 Attributes of GES: foodwebs at species resolution

Creation of species-resolved models foodwebs with macroecological properties comparable to those of temperate shelf communities appears possible only through the process of stepwise assembly from a large species pool described in Section 6.4. This type of community assembly was introduced by Caldarelli *et al*. (1998) and has since been refined (Drossel *et al*., 2001; Yoshida, 2003; Rossberg *et al*., 2008; Shephard *et al*., 2012). Assembly is a slow process. The number of successful invasions required to build a saturated community is on the order of 5–10 time the species richness of the final community. The last species to establish themselves are the largest and highest in the food chain. Their populations also easily collapse under perturbations. This supports the idea that large species should be given high conservation priority.

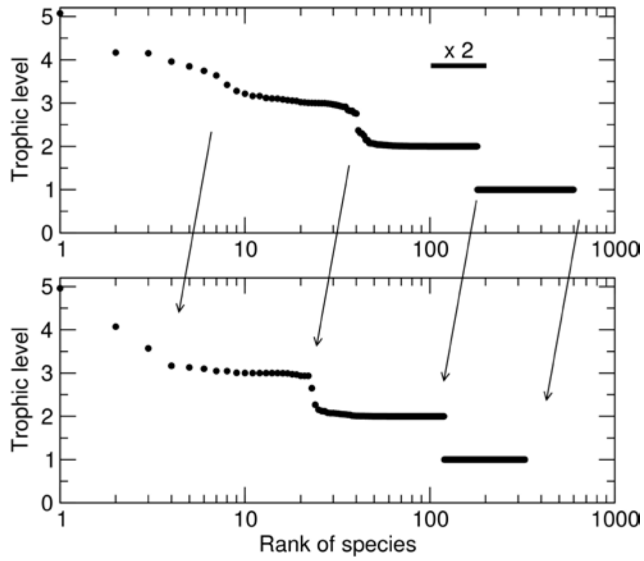

Figure 6.7.4.1. Bottom–up propagation of a loss of species richness through a model foodweb. The upper panel shows the distribution of species over trophic levels in an unperturbed PDMM model foodweb, the lower panel the distribution resulting after randomly removing 50% of producers (trophic level 1) and simulating population dynamics until a new equilibrium state is reached. It turns out that species richness at all trophic levels decreases by about 50%. Each point corresponds to one species, ranked from left to right in order of decreasing trophic level. The scale bar (x 2) indicates the shift on the logscale corresponding by a multiplication or division by 2.

However, the sustainable species richness at higher trophic levels is controlled by that at lower trophic levels. Empirically, species richness declines by a factor of approximately 1/3 with each higher trophic level (Jennings *et al.*, 2002; Petchey *et al.*, 2004). Theory explains this as resulting from bottom–down constraints to coexistence.[9] For example, as demonstrated in Figure 6.7.4.1 using the PDMM (see Section 6.4), removal of 50% of species at the bottom of the food chain leads essentially to 50% biodiversity loss at all higher trophic levels.

Therefore, conservation of biodiversity at the highest trophic levels requires maintenance of biodiversity at all levels. Equal relative changes in species richness at different levels have approximately equal weight. The distribution of species richness over trophic levels is an important attribute of foodwebs.

The inclusion of biodiversity in Descriptor 4 (“at normal abundance and diversity”) is therefore pertinent. As illustrated by Figure 6.7.4.1, the central biodiversity metric for Descriptor 4 is species richness (number of species). The underlying reason is that the range of variability of trophic link strengths among species (~10^6 of more; Rossberg *et al.*, 2011) is much broader than range of variability of biomasses (~10^3). Descriptor 4 concerns a different aspect of biodiversity than Descriptor 1.

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[9] This is an extension to foodwebs of the classical result by Levin (1970) that the number of coexisting consumers cannot be larger than the number or resources. The extension takes the sparse and essentially random structure of foodwebs into account. The implied geometric decay of species richness with trophic level also provides an explanation for the realised lengths of food chains (see Section 6.3).

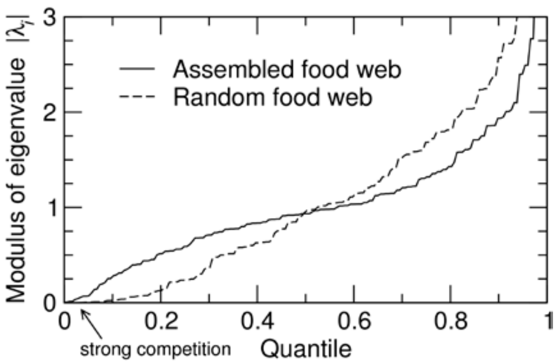

Figure 6.7.4.2. The signature of competition avoidance (species packing) in assembled model communities. Shown is the distribution of the moduli of the eigenvalues λ_i of the competitive overlap matrix α (May, 1975) for the community of all consumers in two foodwebs. Strong competitive interactions are represented by eigenvalues near zero. These make foodwebs sensitive to perturbations. The set of species ultimately sorted out by the process of foodweb assembly (identical with the consumers in Figure 6.7.1, top) leads to fewer eigenvalues near zero than a random set of species with similar numbers of species and trophic links.

Another important aspect of the fine structure of foodwebs seems to be competition avoidance. The main reason why stepwise community assembly is such a prolonged process appears to be that, in the course of this, communities are gradually formed in which competition among resident consumers for resources is weaker than it would be for random foodwebs (Figure 6.7.4.2). It is conceivable that insufficient competition avoidance of this kind will limit species richness (and so production) when communities are reorganized by drivers such as climate change. Dietary overlaps might offer a sufficiently accurate proxy for competitive overlaps (May, 1975) to monitor this attribute. For example, Case (1983) identified signals of competition avoidance in diets of a community of lizards.

6.7.5 Attributes of GES: functional groups

Section 6.5 already introduced models for interactions between functional groups in foodwebs. Whereas the emphasis there was on the role of such models for determining energy fluxes in foodwebs, slow dynamics or bistability in such models are potentially controlled by other variables. For the present purpose, ecological processes do not need to be fully described by such models. They might result, e.g. directly from fitting statistical models for the interactions between functional groups to time-series (e.g. Llope *et al*., 2011).

For the identification of slowly changing variables in models of interacting functional groups, classical techniques based on linearization of system dynamics near an equilibrium point might be sufficient (Luenberger, 1979; Hinrichsen and Pritchard, 2005).

The linearized dynamics are described by the Jacobian matrix J of the model at a fixed point. For each small eigenvalue of J, the corresponding eigenvector defines a slow variable (slow mode) of the system, and so an important attribute of foodweb condition. The eigenvector of the transpose of J for the same eigenvalue (the adjoint eigenvector) specifies the relative sensitivity of the slow variable to pressures on each functional group in the model. The scalar product of the adjoint eigenvector with the deviation of the system state from equilibrium gives the value of the slow variable independent of the remaining system state. Section 6.6 offered an example for obtaining the necessary data on the abundances of functional groups from existing surveys.

Possibly, this set of tools is already sufficient to identify, to monitor, and to manage slow dynamics in foodwebs as represented in models of coupled functional groups.

6.7.6 Summary

The approach developed above to identify important attributes of foodweb GES led to a small, manageable set of attributes. The main criteria for inclusion of attributes in this set were large time-scales for their recovery from pressures and that they can be modelled, so allowing informed management. These criteria made it possible to narrow down the much longer list of characteristics of foodwebs that have been studied in the literature (Bersier, 2007).

6.8 Conclusions and recommendations

The complexity of foodwebs requires a structured approach to developing foodweb condition indicators. Here, we proposed to structure the development in two steps, with the first step consisting in the identification of important attributes of foodwebs, and the second step being the development of indicators matched with the attributes.

Taking the first step, WGECO proposed as important attributes of foodweb functioning:

i) energy flows and transfer efficiencies among functional groups.

As important attributes of foodweb structure were identified:

ii) the amplitudes of the three main responses of community size spectra to pressures;

iii) the distribution of species richness over trophic levels;

iv) the degree of competition avoidance among species;

v) the amplitudes of slow dynamic modes found in models for functional-group interaction.

The next logical step is the identification and, if necessary, the development of indicators addressing each of these aspects; criteria for selecting indicators have been developed under ToR A. At the same time, a critical examination of our proposed list of attributes will be useful. To aid the development of indicators of foodweb function and structure, WGECO request collaboration by other ICES working groups, as detailed I the Annex.

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Appendix 6.1

Trophic guilds to which Northeast Atlantic fish species were assigned based on their ontogenetic switches in diet. (Note that a "Demersal piscivore" is a demersal fish that feeds on fish, not any predator feeding on demersal fish.)

Common name	Scientific name	Length (cm) below	Trophic guild
Anglerfish	*Lophius piscatorius*	0–29	Demersal benthivore
		30–151	Demersal piscivore
Atlantic wolffish	*Anarhichas lupus*	0–4	Pelagic planktivore
		5–150	Demersal benthivore
Ballan wrasse	*Labrus bergylta*	0–1	Pelagic planktivore
		2.0–20	Demersal benthivore
Bib	*Trisopterus luscus*	0–2	Pelagic planktivore
		3.0–20	Demersal benthivore
		21–46	Demersal piscivore
Black anglerfish	*Lophius budegassa*	0–19	Demersal benthivore
		20–100	Demersal piscivore
Black goby	*Gobius niger*	0–1	Pelagic planktivore
		2.0–18	Demersal benthivore
Black-mouthed dogfish	*Galeus melastomus*	0–90	Demersal benthivore
Blonde ray	*Raja brachyura*	0–125	Demersal piscivore
Blue ling	*Molva dypterygia*	0–3	Pelagic planktivore
		4	Demersal benthivore
		5–155	Demersal piscivore
Blue whiting	*Micromesistius poutassou*	0–50	Pelagic planktivore
Bluemouth rockfish	*Helicolenus dactylopterus*	0–1	Pelagic planktivore
		2.0–28	Demersal benthivore
		29–47	Demersal piscivore
Bluntnose Sixgill Shark	*Hexanchus griseus*	0–65	Demersal piscivore
Boar fish	*Capros aper*	0–15	Pelagic planktivore
Bull-rout	*Myoxocephalus scorpius*	0–4	Pelagic planktivore
		0.5–23	Demersal benthivore
		24–60	Demersal piscivore
Butterfish	*Pholis gunnellus*	0–4	Pelagic planktivore
		5.0–30	Demersal benthivore
Cod	*Gadus morhua*	0–3	Pelagic planktivore
		4–250	Demersal piscivore
Common goby	*Pomatoschistus microps*	0–9	Demersal benthivore
Conger eel	*Conger conger*	0–14	Pelagic planktivore
		15–150	Demersal benthivore
Cuckoo ray	*Raja naevus*	0–50	Demersal benthivore
		51–75	Demersal piscivore
Dab	*Limanda limanda*	0–3	Pelagic planktivore
		4.0–40	Demersal benthivore
Dragonet	*Callionymus lyra*	0–1	Pelagic planktivore

Common name	Scientific name	Length (cm) below	Trophic guild
		2.0–30	Demersal benthivore
European pilchard	*Sardina pilchardus*	0–28	Pelagic planktivore
Five-bearded rockling	*Ciliata mustela*	0–2	Pelagic planktivore
		3.0–25	Demersal benthivore
Flounder	*Platichthys flesus*	0–2	Pelagic planktivore
		3.0–60	Demersal benthivore
Four-bearded rockling	*Enchelyopus cimbrius*	0–5	Pelagic planktivore
		6.0–40	Demersal benthivore
Four-spot megrim	*Lepidorhombus boscii*	0–2	Pelagic planktivore
		3.0–40	Demersal benthivore
Garfish	*Belone belone*	0–93	Demersal benthivore
Goldsiny wrasse	*Ctenolabrus rupestris*	0–1	Pelagic planktivore
		2.0–18	Demersal benthivore
Greater argentine	*Argentina silus*	19–70	Pelagic planktivore
Greater forkbeard	*Phycis blennoides*	0–3	Pelagic planktivore
		4.0–21	Demersal benthivore
		22–110	Demersal piscivore
Greater sandeel	*Hyperoplus lanceolatus*	0–3	Pelagic planktivore
		4.0–40	Demersal piscivore
Grey gurnard	*Eutrigla gurnardus*	0–3	Pelagic planktivore
		4.0–27	Demersal benthivore
		28–60	Demersal piscivore
Haddock	*Melanogrammus aeglefinus*	0–2	Pelagic planktivore
		3.0–59	Demersal benthivore
		60–112	Demersal piscivore
Hake	*Merluccius merluccius*	0–2	Pelagic planktivore
		3.0–20	Demersal benthivore
		21–140	Demersal piscivore
Halibut	*Reinhardtius hippoglossoides*	0–3	Pelagic planktivore
		4.0–14	Demersal benthivore
		15–80	Demersal piscivore
Herring	*Clupea harengus*	0–45	Pelagic planktivore
Hollow snout grenadier	*Caelorinchus caelorinchus*	0–5	Pelagic planktivore
		6.0–48	Demersal piscivore
Hooknose	*Agonus cataphractus*	0–2	Pelagic planktivore
		3.0–21	Demersal benthivore
Horse mackerel	*Trachurus trachurus*	0–25	Pelagic planktivore
		26–70	Pelagic piscivore
Imperial scaldfish	*Arnoglossus imperialis*	0–2	Pelagic planktivore
		3.0–25	Demersal benthivore
John Dory	*Zeus faber*	0–8	Pelagic planktivore
		9.0–90	Demersal piscivore
Lanternfish	*Myctophum punctatum*	0–11	Pelagic planktivore

Common name	Scientific name	Length (cm) below	Trophic guild
Lemon sole	*Microstomus kitt*	0–4	Pelagic planktivore
		5.0–65	Demersal benthivore
Lesser sandeel	*Ammodytes tobianus*	0–1	Pelagic planktivore
		2.0–20	Demersal benthivore
Lesser spotted dogfish	*Scyliorhinus canicula*	0–100	Demersal benthivore
Ling	*Molva molva*	0–30	Pelagic planktivore
		31–200	Demersal piscivore
Long rough dab	*Hippoglossoides platessoides*	0–3	Pelagic planktivore
		4.0–70	Demersal benthivore
Lumpsucker	*Cyclopterus lumpus*	0–5	Pelagic planktivore
		6.0–61	Demersal benthivore
Mackerel	*Scomber scombrus*	0–25	Pelagic planktivore
		26–60	Pelagic piscivore
Megrim	*Lepidorhombus whiffiagonis*	0–2	Pelagic planktivore
		3.0–60	Demersal benthivore
Moustache sculpin	*Triglops murrayi*	0–2	Pelagic planktivore
		3.0–40	Demersal benthivore
Norway pout	*Trisopterus esmarki*	0–6	Pelagic planktivore
		7.0–35	Demersal benthivore
Pearlside	*Maurolicus muelleri*	0–8	Pelagic planktivore
Plaice	*Pleuronectes platessa*	0–2	Pelagic planktivore
		3.0–100	Demersal benthivore
Pollack	*Pollachius pollachius*	0–24	Pelagic planktivore
		25–81	Demersal benthivore
Poor -cod	*Trisopterus minutus*	0–3	Pelagic planktivore
		4.0–30	Demersal benthivore
Rabbit/ratfish	*Chimaera monstrosa*	0–150	Demersal benthivore
Red band fish	*Cepola rubescens*	0–3	Pelagic planktivore
		4.0–80	Demersal benthivore
Redfish	*Sebastes marinus*	0–24	Pelagic planktivore
		25–59	Demersal benthivore
		60–100	Demersal piscivore
Rock gurnard	*Trigloporus lastoviza*	0–1	Pelagic planktivore
		2.0–40	Demersal benthivore
Round skate	*Rajella fyllae*	0–60	Demersal benthivore
Roundnose grenadier	*Coryphaenoides rupestris*	0–1	Pelagic planktivore
		2.0–110	Demersal benthivore
Saithe	*Pollachius virens*	0–45	Pelagic planktivore
		46–10	Demersal piscivore
Salmon	*Salmo salar*	0–25	Pelagic planktivore
		26–150	Pelagic piscivore
Sand goby	*Pomatoschistus minutus*	0–11	Demersal benthivore
Sand sole	*Pegusa lascaris*	0–2	Pelagic planktivore

Common name	Scientific name	Length (cm) below	Trophic guild
		3.0–40	Demersal benthivore
Scaldfish	*Arnoglossus laterna*	0–1	Pelagic planktivore
		2.0–25	Demersal benthivore
Sea scorpion	*Taurulus bubalis*	0–1	Pelagic planktivore
		2.0–60	Demersal benthivore
Sea snail	*Liparis montagui*	0–2	Pelagic planktivore
		3.0–12	Demersal benthivore
Sea trout	*Salmo trutta*	0–15	Pelagic planktivore
		16.0–140	Pelagic piscivore
Small-eyed ray	*Raja microocellata*	0–64	Demersal benthivore
		65–91	Pelagic piscivore
Smooth hound	*Mustelus mustelus*	0–164	Demersal benthivore
Sole	*Solea solea*	0–2	Pelagic planktivore
		.0–70	Demersal benthivore
Solenette	*Buglossidium luteum*	0–1	Pelagic planktivore
		2.0–15	Demersal benthivore
Spotted Dragonet	*Callionymus maculatus*	0–2	Pelagic planktivore
		3.0–16	Demersal benthivore
Spotted ray	*Raja montagui*	0–50	Demersal benthivore
Spotted wolfish	*Anarhichas minor*	0–4	Pelagic planktivore
		5.0–100	Demersal benthivore
		101–150	Demersal piscivore
Sprat	*Sprattus sprattus*	0–16	Pelagic piscivore
Spurdog	*Squalus acanthias*	0–60	Demersal benthivore
		61–120	Demersal piscivore
Starry ray	*Raja radiata*	0–41	Demersal benthivore
		42–100	Demersal piscivore
Starry smooth hound	*Mustelus asterias*	0–140	Demersal benthivore
Stingray	*Dasyatis pastinaca*	0–100	Demersal benthivore
Striped red mullet	*Mullus surmuletus*	0–5	Pelagic planktivore
		6.0–21	Demersal benthivore
Thickback sole	*Microchirus variegatus*	0–2	Pelagic planktivore
		3.0–35	Demersal benthivore
Thornback ray	*Raja clavata*	0–50	Demersal benthivore
		51–139	Demersal piscivore
Thorny skate	*Amblyraja radiata*	0–105	Demersal benthivore
Three-bearded rockling	*Gaidroparus vulgaris*	0–5	Pelagic planktivore
		6.0–60	Demersal benthivore
Three-bearded rockling	*Gaidropsarus vulgaris*	0–2	Pelagic planktivore
		3.0–60	Demersal benthivore
Tope	*Galeorhinus galeus*	0–193	Demersal piscivore
Turbot	*Psetta maxima*	0–3	Pelagic planktivore
		4.0–22	Demersal benthivore

Common name	Scientific name	Length (cm) below	Trophic guild
		23.0–82	Demersal piscivore
Tusk	*Brosme brosme*	0–5	Pelagic planktivore
		6.0–100	Demersal piscivore
Velvet belly latern shark	*Etmopterus spinax*	0–20	Pelagic planktivore
		21–60	Demersal piscivore
Whiting	*Merlangius merlangus*	0–5	Pelagic planktivore
		6.0–20	Demersal benthivore
		21.0–70	Demersal piscivore
Witch	*Glyptocephalus cynoglossus*	0–6	Pelagic planktivore
		7.0–60	Demersal benthivore

7 ToRe) LFI development

The Large Fish indicator (LFI) was originally developed to support the OSPAR 'fish community' EcoQO for the North Sea (WGECO, 2010; Greenstreet *et al*., 2011). During the ICES/JRC process to select a set of indicators for the "Marine Strategy Framework Directive" (MSFD), the LFI was proposed as a potential Descriptor 4 'FoodWeb' indicator (EC, 2008). Since the LFI had been developed to express change in fish community size structure, WGECO (2011) suggested that further research was required to clarify if it had genuine utility as a foodweb indicator. Subsequent work has explored the function of this indicator and its application in other marine regions (e.g. Greenstreet *et al*., 2012; Shephard *et al*., 2011ab; 2012; Fung *et al*., 2012; Heath and Speirs, 2011). WGECO 2012 extended this research, (1) tuning the indicator for the Baltic Sea and the Gulf of Cadiz, (2) exploring the potential effects of interaction between different sampling gears and fish communities and (3) considering the application of the LFI in MSFD sub regions which are sampled by several independent fisheries surveys.

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7.1 Comparison of LFI trends based on the North Sea beam trawl and otter trawl surveys

7.1.1 Introduction

The LFI is the basis for the Ecological Quality Objective (EcoQO) of the North Sea fish community, and a single management target has been set for this whole OSPAR region, based on the data collected during the IBTS in Q1 (OSPAR, 2009). However, diverse studies have shown the spatial variability of the fish community within the North Sea (Daan *et al*., 1990; Fraser *et al*., 2008). This variability is reflected in the spatial distribution of fishing effort with different gears, beam trawling targeting flatfish being more common in the southern North Sea and otter trawling targeting roundfish dominating in the northern North Sea (Jennings *et al*., 1999).

WGECO compared LFI trends obtained from the beam trawl and otter trawl surveys datasets with two aims; first, to investigate the existence of regional differences in the LFI trends obtained with each gear as opposite to the estimation of a single LFI trend for the whole North Sea; second, to investigate if there are differences in the LFI trends estimated from surveys using different gears.

7.1.2 Material and methods

Two datasets were used in this analysis; the North Sea Beam Trawl Survey (BTS) and the International Bottom-trawl Survey, which uses an otter bottom trawl. Because the Beam Trawl Survey is carried out in Q3, the data used in this analysis from the International Bottom-trawl Survey were restricted to this same quarter. Both datasets were downloaded from the ICES Datras database and split into northern and southern regions, using the same SW–NE dividing line as in Fraser *et al*. (2008). The resulting northern and southern North Sea subsets included data from 63 and 55 ICES statistical rectangles, respectively.

The Beam Trawl Survey dataset spanned the period 1990–2011 for the southern basin but 1996–2011 for the northern region, thus the data prior to 1996 were not used in the final analysis. The total number of valid hauls from the Beam Trawl Survey dataset used in the analysis was 702 for the northern region, split by gear as follows: 90 hauls taken with a BT7 gear and 612 with a BT8. For the southern North Sea region, data from 2633 valid hauls were available: 252 taken with BT4, 260 with BT7 and 2121 with BT8. Due to time constraints and since the majority of hauls were taken with beam trawls of similar size, no effort was made to estimate separately biomass per hour at length for the BT4.

The available data from the Beam Trawl Survey were not enough to estimate swept-area and data were thus standardized as biomass per hour. For consistency, the same was done with the International Bottom-trawl Survey data. This dataset covers the period 1991–2011 in the northern and southern basin, and included 3553 valid hauls in the northern region and 2102 valid hauls in the southern region of the North Sea.

We followed the protocol described in Greenstreet and Rogers (2006) to estimate the LFI indices; and to make our results comparable to those obtained from the otter trawl survey data, we tested the threshold fish lengths of 30, 35 and 40 cm.

The analysis was carried out in R (R Development Core Team, 2011), using the R DatrasR package written by D. Beare.

7.1.3 Results

7.1.3.1 Analysis of Beam Trawl Survey data

LFI trends from the northern North Sea and southern North Sea differed greatly, decreasing in the northern North Sea but increasing in the southern North Sea. When data from both regions were combined, it became apparent that the southern North Sea subset weighted more in the combined LFI. The range of values for all data combined was intermediate to the range obtained for both datasets separately, but the trend was the same as for the southern North Sea subset (Figure 7.1.3.1). The best fit was obtained for the 30 cm threshold in both regions and for the whole North Sea.

Beam trawl (BTS), Quarter 3

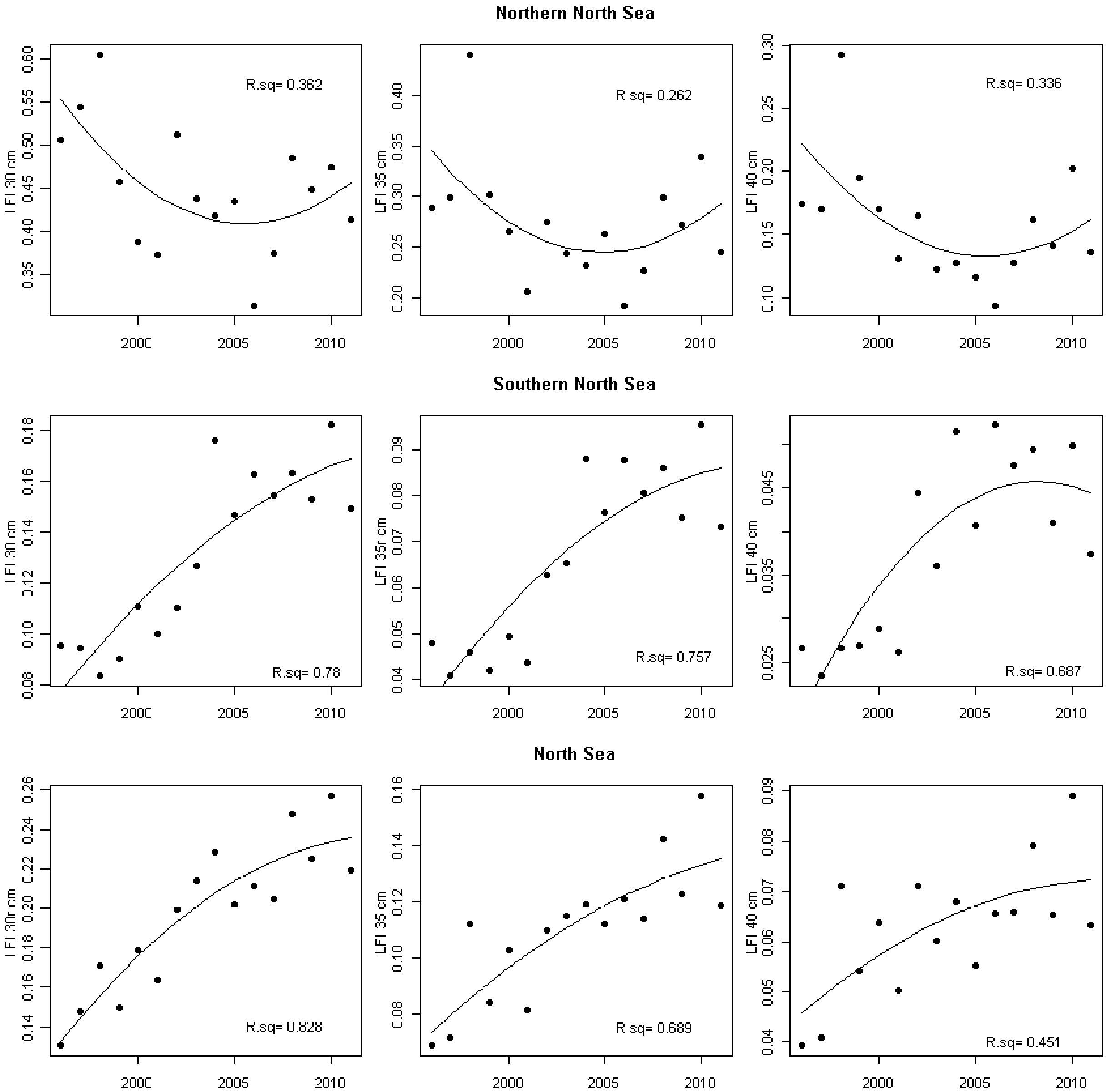

Figure 7.1.3.1. LFI trends estimated from the BTS survey dataset and 4th degree polynomial fits for the regional subsets and whole North Sea at three different size thresholds, 30, 35 and 40 cm.

Flatfish dominated catches in both the northern and southern regions considered in this analysis, representing >50% and >85% of the biomass per hour, respectively. Gadoids were more common in the northern region (about 20%) than in the southern (about 5%), and they were mostly haddock, cod and whiting. The most abundant flatfish species were plaice and dab (Figures 7.1.3.2 and 7.1.3.3).

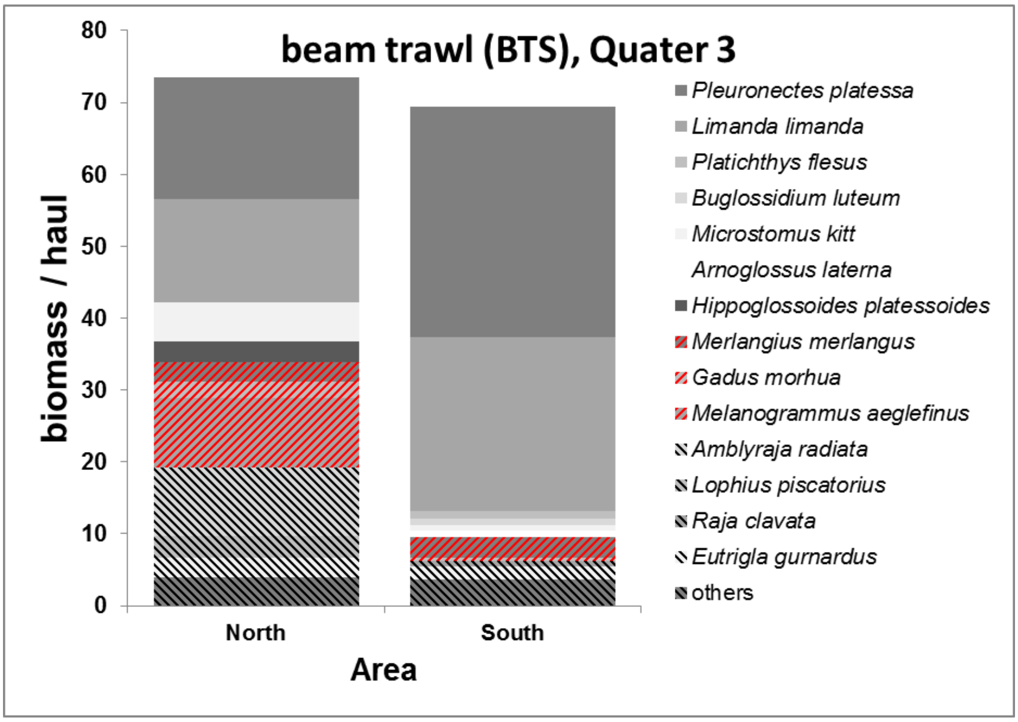

Figure 7.1.3.2. Biomass (kg) per haul / hour of the most common species registered in the beam trawl survey in the northern (North) and southern (South) North Sea. Gadoids marked with overlapping red lines, flatfish species in plain colour and other species marked with overlapping black lines.

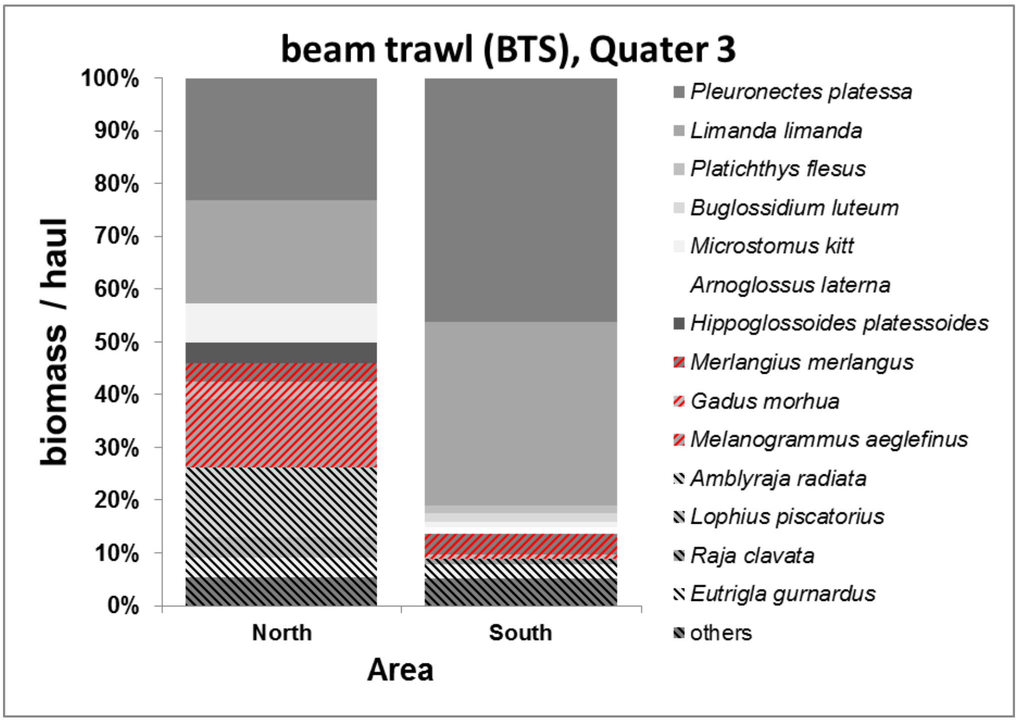

Figure 7.1.3.3. Proportion of biomass (kg) per haul/hour of the most common species registered in the beam trawl survey in the northern (North) and southern (South) North Sea. Gadoids marked with overlapping red lines, flatfish species in plain colour and other species marked with overlapping black lines.

7.1.3.2 Analysis of the International Trawl Survey data

Again, data from the northern and southern North Sea showed very different LFI trends. Furthermore, within each region, the LFI trends obtained from the otter trawl data were opposite to those obtained with the beam trawl dataset: increasing in the northern North Sea but decreasing in the southern North Sea. After we combined the otter trawl data from both regions, the northern North Sea subset weighted more in the combined LFI. Both the range of values and the trends were the same as for the northern North Sea subset (Figure 7.1.3.4). The best fit was obtained for the 30 cm threshold in the southern North Sea and for the whole North Sea, whereas r-squared was nearly the same in the northern North Sea for the 30 and 35 cm thresholds.

Otter trawl (IBTS), Quarter 3

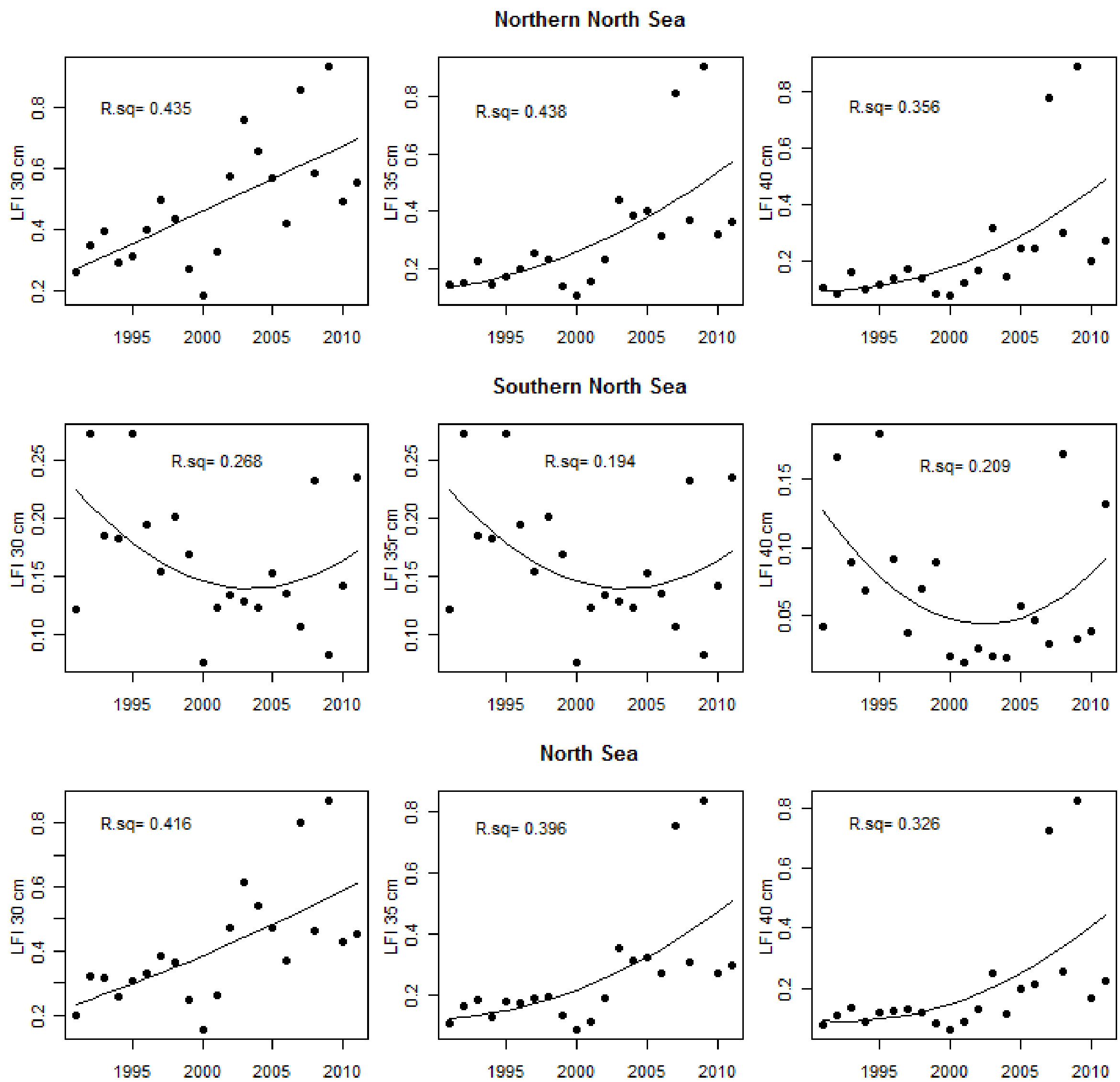

Figure 7.1.3.4. LFI trends estimated from the IBTS survey dataset and 4th degree polynomial fits for the regional subsets and whole North Sea at three different size thresholds, 30, 35 and 40 cm.

Caught biomass per haul/hour with the otter trawl was nearly twice as much in the northern region than in the southern North Sea. Gadoids were the dominant group, representing roughly 70% of the catch in the northern North Sea and 50% in the southern North Sea. Among gadoids, the most abundant species were haddock and whiting. Flatfish represented <10% in the northern North Sea but 30% in the southern North Sea, and dab was the most abundant flatfish species in the otter trawl catch (Figures 7.1.3.5 and 7.1.3.6).

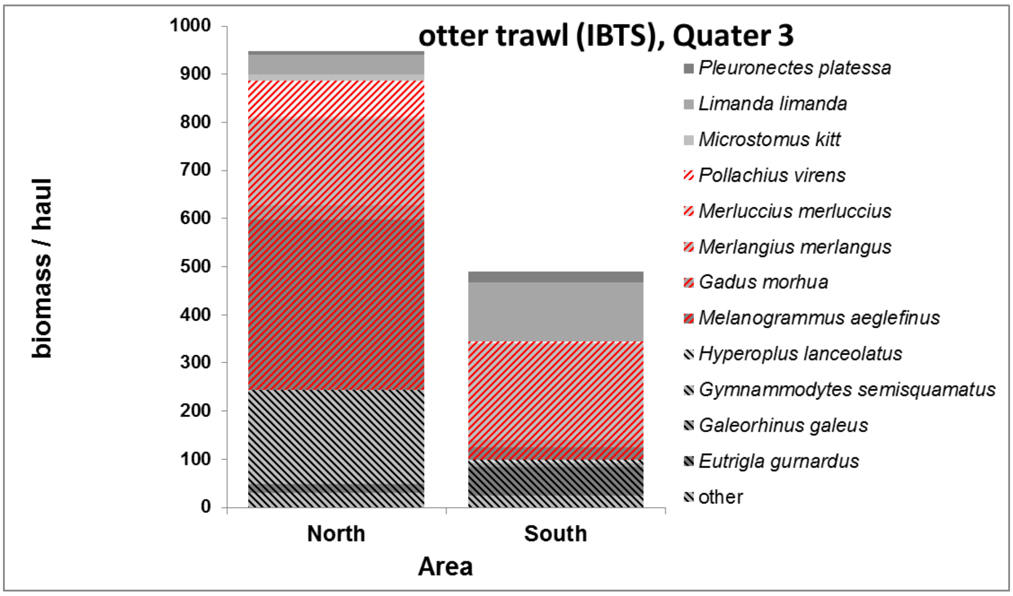

Figure 7.1.3.5. Biomass (kg) per haul/hour of the most common species caught in the IBTS, Q3, during the period 1991–2011.

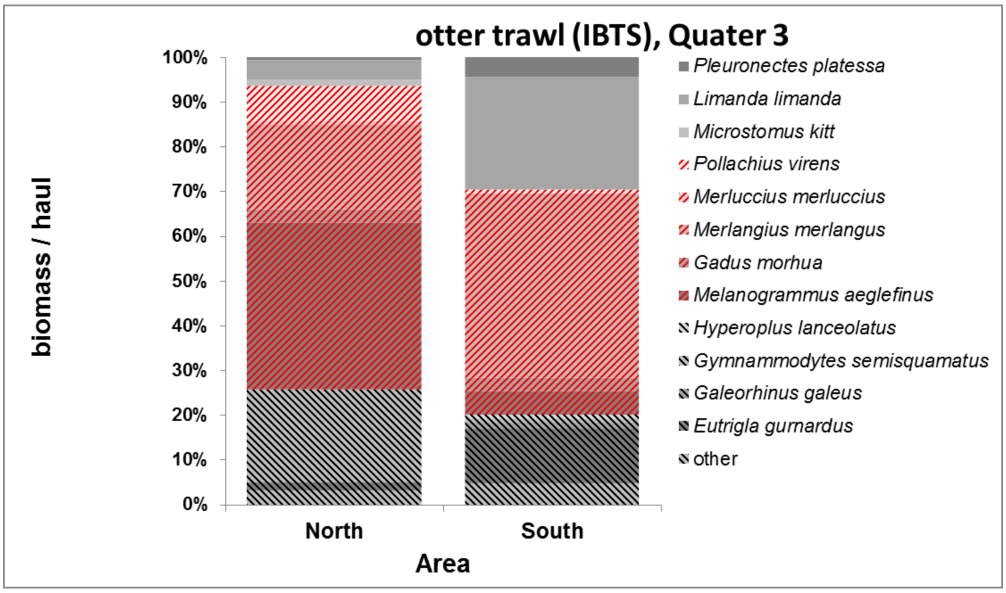

Figure 7.1.3.6. Proportion of biomass (kg) per haul/hour of most common species caught in the IBTS, Q3, during the period 1991–2011.

7.1.4 Discussion

Our results suggest that the LFI index is sensitive to gear selectivity and regional variability of the fish assemblage.

Previously reported differences in the fish assemblage between the northern and southern regions of the North Sea (e.g. Fraser *et al*., 2008) were evident in the current analysis; more gadoid species were typically caught in the northern area and more flatfish were caught in the southern area. However, the relative importance of gadoids vs. flatfish in catch composition within each area differed between gears (Table 7.1.4.1).

Table 7.1.4.1. Percentage of catch of gadoids and flatfish with different gears in the northern and southern North Sea regions.

	N			S		
	Gadoids	Flatfish	Other	Gadoids	Flatfish	Other
Beam Trawl	20	55	25	5	85	10
Otter Trawl	70	5	25	50	30	20

Interaction between sampling gear and fish assemblage may be partly responsible for the different range of indicator values, which are higher for the LFI obtained from beam trawl data. Trends in the LFI series for north and south areas also differed between gears. The LFI in the northern North Sea showed a clearly increasing trend for the whole study period when estimated from otter trawl data, but when estimated from beam trawl data it showed a decrease in the early years. In the southern region, the otter trawl data showed a variable trend, but beam trawl data show a clear increasing trend.

The reason for these observed differences in indicator series becomes clear when comparing series for northern and southern LFIs for each survey to the corresponding overall survey LFI. When the beam trawl data were pooled to estimate a single LFI for the whole North Sea, the indicator time-series was very similar to that derived for the area in which this gear caught greater biomass and sampled the fish assemblage most efficiently (southern). Similarly, the overall North Sea LFI series calculated for the otter trawl survey appeared to be driven predominately by the northern area (where this gear samples the (gadoid) assemblage most efficiently and most of the survey biomass is caught). The apparent likelihood of confounding sampling gear with fish assemblage suggests that surveys should be disaggregated by biogeographic region.

7.1.5 References

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7.2 Combining multiple surveys to derive regional scale assessments of the status of fish communities from subregional scale datasets

The Marine Strategy Framework establishes four European Marine Regions, based on geographical and environmental criteria. Each Member State is required to develop a marine strategy for their own waters (EEZs or extended continental shelf areas), but Good Environmental Status will be considered at regional or subregional scale. The Northeast Atlantic Marine Region is divided into four subregions (Atlantic Ocean; Bay of Biscay and the Iberian coast; Celtic Seas and Greater North Sea). The spatial extent and ecological diversity of the subregions presents a challenge for data collection and monitoring programmes, especially for widespread and diverse ecosystem components such as fish communities. In the North Sea, fish community monitoring is strongly facilitated by a well-integrated international bottom-trawl survey that covers much of the 'Greater North Sea' subregion. However, in other subregions, fisheries surveys are typically conducted by individual nations, with spatial coverage referring to national waters or traditional fishing grounds. The Celtic Seas subregion is covered by at least three separate surveys, which occur in different (but sometimes overlapping) areas and seasons and follow somewhat differing protocol. WGECO used the protocol of Greenstreet *et al*. (2011) and Shephard *et al*. (2011) to derive 'optimal' LFI definitions for each Celtic Sea survey (disaggregated by biogeographic region). Time-series of these LFIs were then visually compared (Figure 7.2.1).

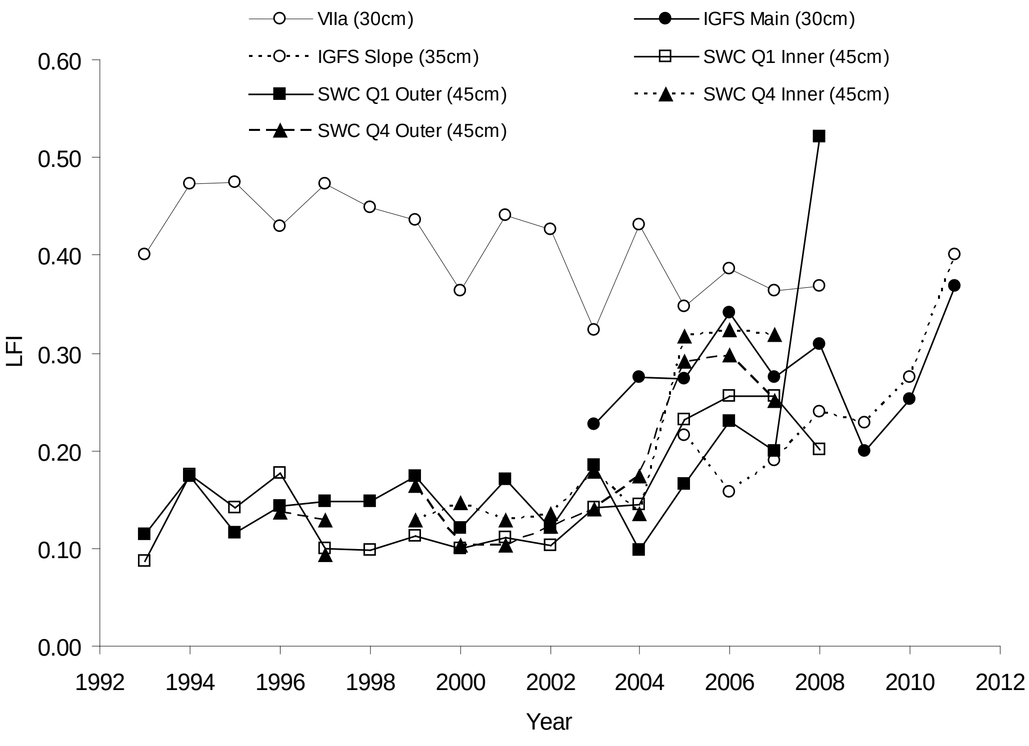

Figure7.2.1. LFI series derived from Celtic Sea survey biogeographic regions.

Comparison suggests that there are some differences in LFI series derived from different surveys and survey biogeographic areas in the Celtic Seas subregion. Specifically, the IGFS and SWC surveys show LFI values in the same range whereas the Irish Sea LFI shows markedly different values and trend. In this context, some objective means of creating an ensemble subregion LFI metric is probably required. The precautionary principle suggests that GES for the Celtic Sea fish community should not be assumed unless all component parts of this subregion metric (including the Irish Sea) attain stated management thresholds.

7.3 Examples of regional LFI series

7.3.1 An LFI in the Gulf of Cádiz (SW Spain)

7.3.1.1 Introduction and methods

The Gulf of Cádiz is located in the ICES IXa area. Due to the confluence of the Atlantic Ocean and the Mediterranean Sea (Figure 7.3.1) this area supports a high biodiversity in fauna and flora.

The trawl fleet in the Gulf of Cádiz is quite homogenous, with larger trawlers divided in two main groups. The larger group works in shallow waters (30–100 m), targeting a multsipecific fishery of sparids, cephalopods, wedge sole, hake and horse mackerel. The other group operates within a depth range of 90–500 m, and targets mainly blue whiting, shrimp, horse mackerel, hake and Norway lobster. Hake landings make up around 7–8% of the total landings of this fleet. The rest of the landings are very heterogeneous and include more than 30 different species (fish, shrimp and cephalopods) (WGHMM 2010 report).

An exploratory analysis was carried out to calculate the LFI in the Gulf of Cádiz. The LFI was determined using cpue data disaggregated by length of species from bottom-trawl surveys (ARSA). Regional weight-at-length relationships were used to convert abundance-at-length to weight-at-length. Different length thresholds were used to define "large" fish; 25 cm, 30 cm, 35 cm and 40 cm. In the North Sea the metric relates only to the demersal fish community and pelagic species are excluded from the analysis. The analysis of Gulf of Cádiz data examined two options, firstly calculating the indicator with all demersal species and including pelagic species with demersal behaviour (*Micromesistius poutassou*, *Trachurus trachurus* and *Trachurus mediterraneus*), and secondly, excluding all pelagic species.

Information about species composition is shown to better understand LFI performance in this region. The variability of fishing pressure is also analysed in relation to LFI. The estimation of fishing mortality was based on hake data reported on WGHMM (2011 report):

$$Fcd = Ftot * Ccd / Ctot$$

where Fcd is fishing mortality in the study area, Ftot is the fishing mortality for Divisions VIIc and XIa (as used in the assessment), Ccd is the catch in the study area and Ctot is total catch in divisions VIIc and XIa. Areas. It is assumed that both hake length distribution and density (number of individuals/division) within the study area were the same as in Divisions VIIc and XIa.

7.3.1.2 Survey details

The survey on the Southern Spanish shelf (ARSA) was designed for evaluation of demersal resources (biomass and abundance indices) using a random stratified sampling scheme. The survey started in 1993 and is carried out in both spring and autumn on board of RV Cornide Saavedra, being interrupted in 2003 and in autumn 1994, 1995 and 1996. For this reason, this exploratory analysis used data from spring surveys alone. The gear used is a Baka 44/60 bottom-trawl gear with a 60.3 m headline and 43.8 m footrope. It has a stretched 40 mm mesh in the codend and internally covered with a 20 mm mesh. Mean vertical openings were 1.8 and 2.1 m, respectively. The sampling unit consisted of 1 hour hauls during daytime at a 3 knots speed. The depth range surveyed is 30–800 m.

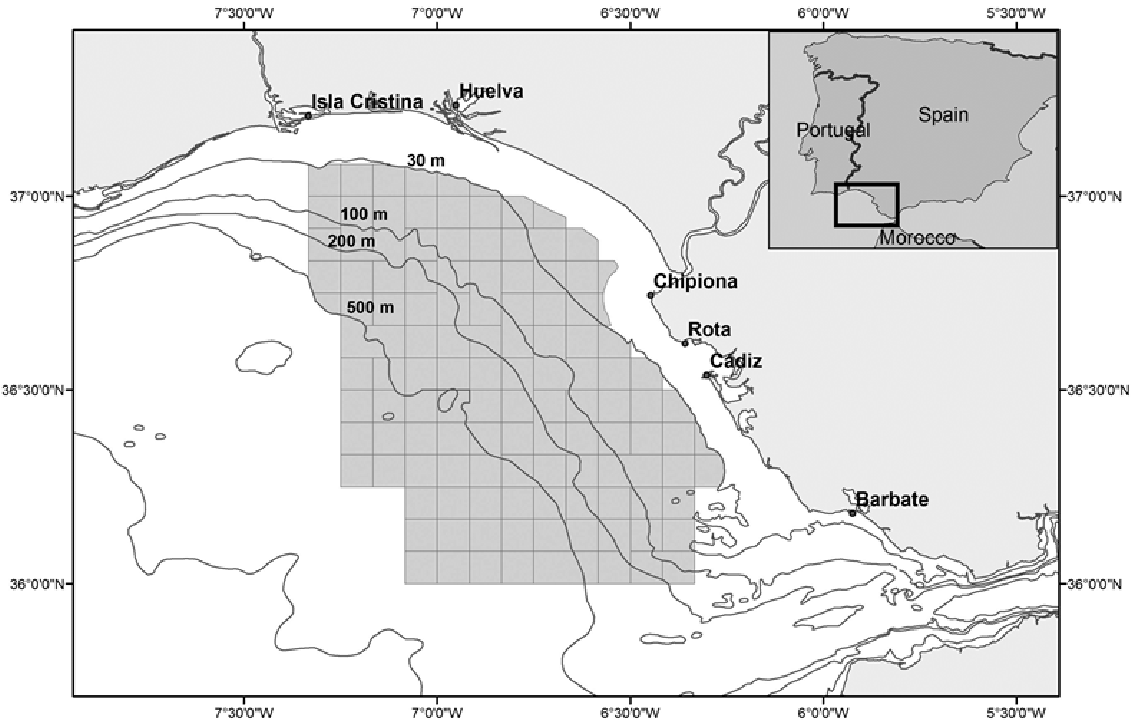

Figure 7.3.1. Map of the area covered by ARSA surveys. Grey grids show sampling locations.

7.3.1.3 Results

Separate analysis of LFI using different thresholds did not substantially affect trends in the LFI.

The best smoother fit was found with a 6th order polynomial for threshold 25 cm, with $r^2 = 0.65$ (including some pelagic species) and $r^2 = 0.54$ (excluding pelagic species) (Figure 7.3.1).

In the WGECO report (2010) it was suggested that the 40 cm threshold might be suitable for defining South Western Waters RAC region, but defining a suitable reference period and setting targets or limits for the LFI needed further work. A recent study from the Northern Spanish shelf revised these data and proposed 35 cm as the most suitable threshold for the LFI calculation. For the Mediterranean Sea (Spanish shelf) the most suitable threshold was also analysed and a 30 cm threshold was proposed (personal communication, Preciado I.). According to this information it seemed unrealistic to choose a threshold size below those for an Atlantic area. However, the different methods tested to define large fish suggested that the indicator performed better at 25 cm and when some pelagic species of demersal behaviour (*Micromesistius poutassou*, *Trachurus trachurus* and *Trachurus mediterraneus*) were included (r2 = 0.65). This threshold size was therefore chosen for the exploratory analysis on species composition (Figure 7.3.2).

The large fish component (≥25 cm) was composed by 21 species, whereas the small fish component (<25 cm) includes 40 species. Biomass trends suggest that changes in the large fish component could be mostly attributed to four dominant species (*Galeus melastomus*, *Trachurus trachurus*, *Scyliorhinus canicula* and *Merluccius merluccius*) whereas the small fish component was dominated by *Trachurus trachurus*, *Micromesistius poutassou*, *Merluccius merluccius* and *Boops boops* (Figure 7.3.3). The LFI seemed to be mostly influenced by changes in the fish <25 cm long biomass. Specifically, the decline in LFI during the periods 1993–1994 and 1998–2004 coincided with high values of biomass of the small fish component (*Trachurus trachurus* and *Micromesistus poutassou*). Fishing mortality and LFI showed also an opposite trend for the period 1998–2004 (Figure 7.3.4). From 2005 an increasing trend on LFI was observed as fishing mortality decreased. This is an exploratory analysis and further work is required.

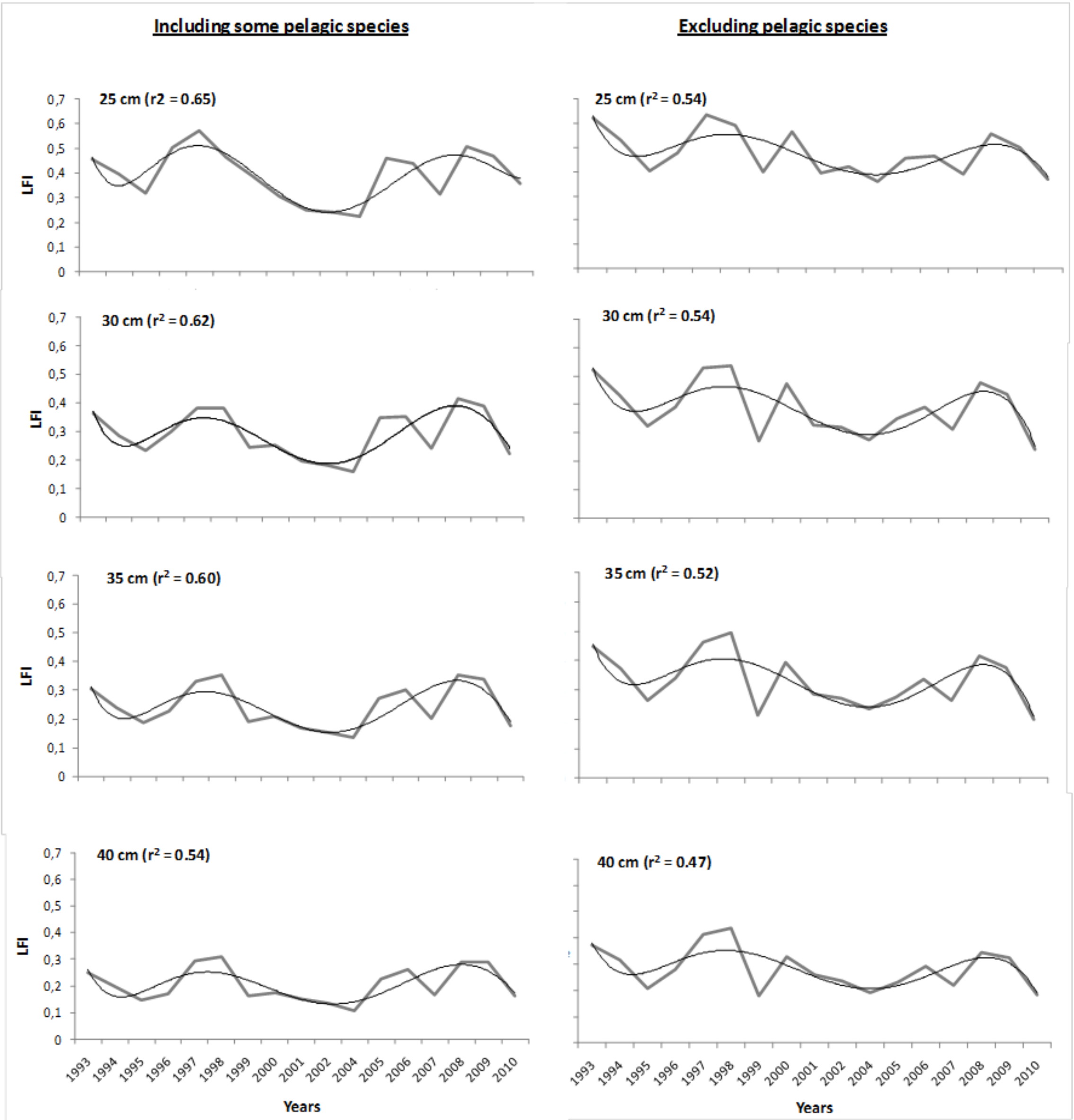

Figure 7.3.2. LFI variation using four different length thresholds (25, 30, 35 and 40 cm) and two options: including some pelagic species (*Micromesistius poutassou*, *Trachurus trachurus* and *Trachurus mediterraneus*) or excluding all pelagic species. The fitted line is a smoother (6 h order polynomial) with fit (r2) given for each line.

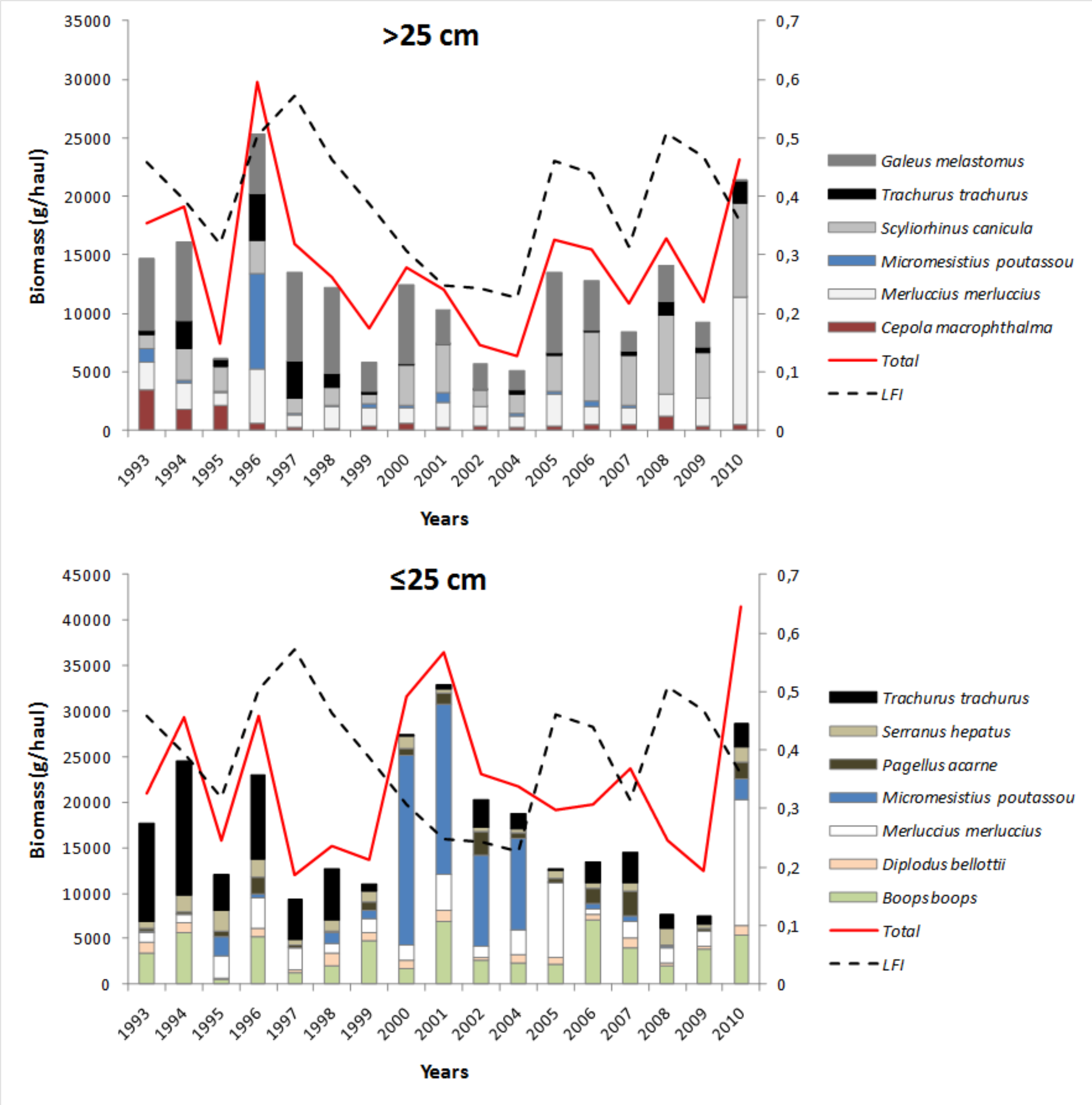

Figure 7.3.3. Variations in standardized biomass of the dominant demersal species and including some pelagic species (*Micromesistius poutassou*, *Trachurus trachurus* and *Trachurus mediterraneus*).

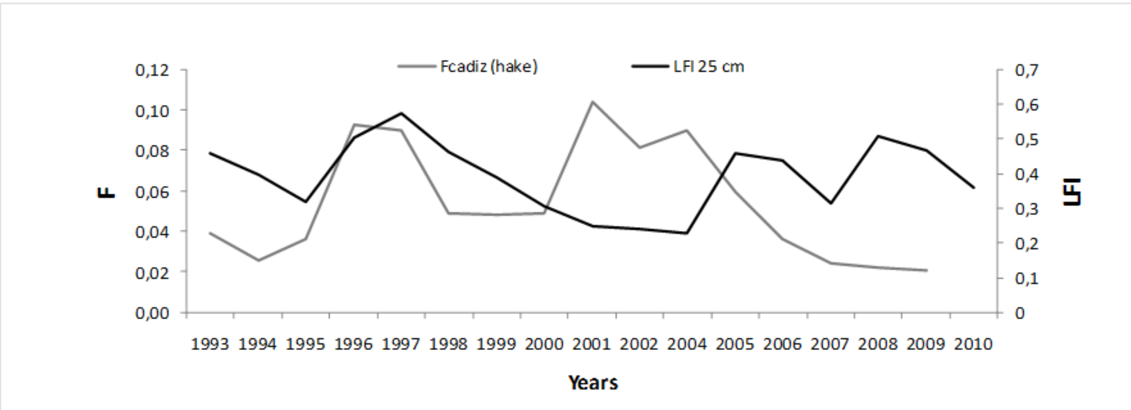

Figure 7.3.4. Trends in fishing mortality (*Merluccius merluccius*) and the LFI (25 cm) where the demersal fish assemblage includes some pelagic species (*Micromesistius poutassou*, *Trachurus trachurus* and *Trachurus mediterraneus*).

7.3.2 Baltic Sea

7.3.2.1 Introduction

Due to the influence of brackish water, species diversity in the Baltic Sea is lower than in many marine systems, and the length of common marine fish species is smaller compared to their relatives living in saltier waters. Together these ecological differences demand some modification of the LFI as developed in the North Sea. Last year we started a preliminary study for the application of the Large Fish Indicator for the Baltic Sea (ICES, 2011a). Further analyses and modifications were done in this study.

7.3.2.2 Method

A BITS (Exchange-) dataset from quarter 1, hold at the ICES database (ICES, 2011b), was downloaded in September 2011. After combining the tables of the haul information and length–frequency, data were sorted according to the ICES Subdivisions 22, 23, 24 and catch per unit of effort data (cpue; number per hour) were calculated. For subsequent analysis only data from those hauls were used, where all bycaught fish species were recorded. Due to the focus of BITS on demersal fish species and the use of bottom-trawl gears, all pelagic species were excluded from the analysis.

Based on the available data, we choose nine commercial fish species which represent 98% of the total biomass caught (Table 1). To calculate the LFI, data from 1991 to 2011 were combined and length–weight relationships were calculated for each fish species. Catch numbers per hour at length were converted to weight-at-length using the calculated weight–length relationships.

Table 7.3.2.1 Proportion of the nine selected species of the summed total biomass of the BITS catches from 1991 to 2011 in the western Baltic (ICES Subdivision22–24).

Species	Proportion of total biomass (%)
Gadus morhua	61.69
Limanda limanda	12.28
Platichthys flesus	11.15
Merlangius merlangus	7.95
Pleuronectes platessa	4.99
Psetta maxima	0.52
Scophthalmus rhombus	0.14
Solea solea	0.03
Merluccius merluccius	0.03
Total	98.77

The sum of biomass of individuals, grouped in three size classes: fish larger than 20 cm, larger than 30 cm and larger than 40 cm, was calculated. For every year, these sums were divided each by the total biomass of the nine species caught in the survey, resulting in three different LFIs, ($LFI_{>20cm}$, $LFI_{>30cm}$, $LFI_{>40cm}$).

7.3.2.3 Results

The nine selected fish species represented more than 98% of the total biomass of the BITS catches between 1991 and 2011.

However, cod is dominating the total biomass of the nine species independently of the length classes (Figure 7.3.5). The larger the length class the more prominent the biomass of cod.

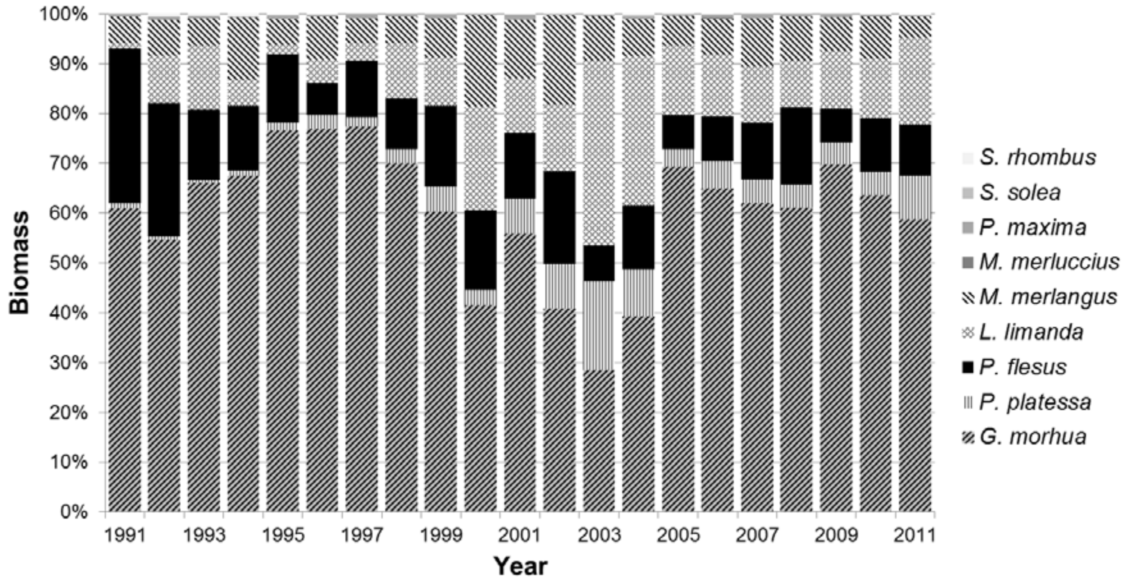

Figure 7.3.5. Proportion of total biomass of the selected fish community for the whole time-series.

The three different LFIs exhibit the same long-term trend (Figure 7.3.6) but small differences were observable (e.g. 2003 and 2004).

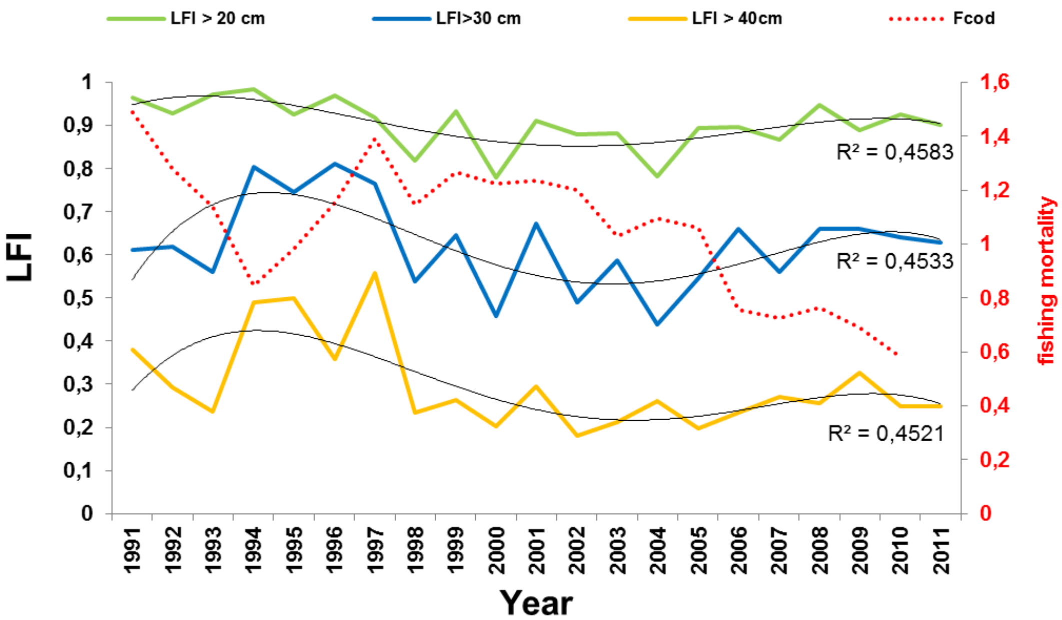

Figure 7.3.6. Time-series of three different 'Large Fish Indicators' (LFIs) in the Western Baltic between 1991 and 2011 and fishing mortality of cod.

The former study was based on four species, while the current study takes nine species into account. Due to that, a comparison of the overall trend between the study carried out last year (Figure 7.3.7; ICES, 2011) and this study shows differences for each LFI length threshold. As an example, the former study shows an increase over the last years, the current study show a decrease or a stable trend. Especially the LFI with a 20 cm and 30 cm threshold differ. This could be explained by the fact that the cod biomass, which was also included last year, is strongly dominating the LFI of 40 cm.

But still further analyses have to be done.

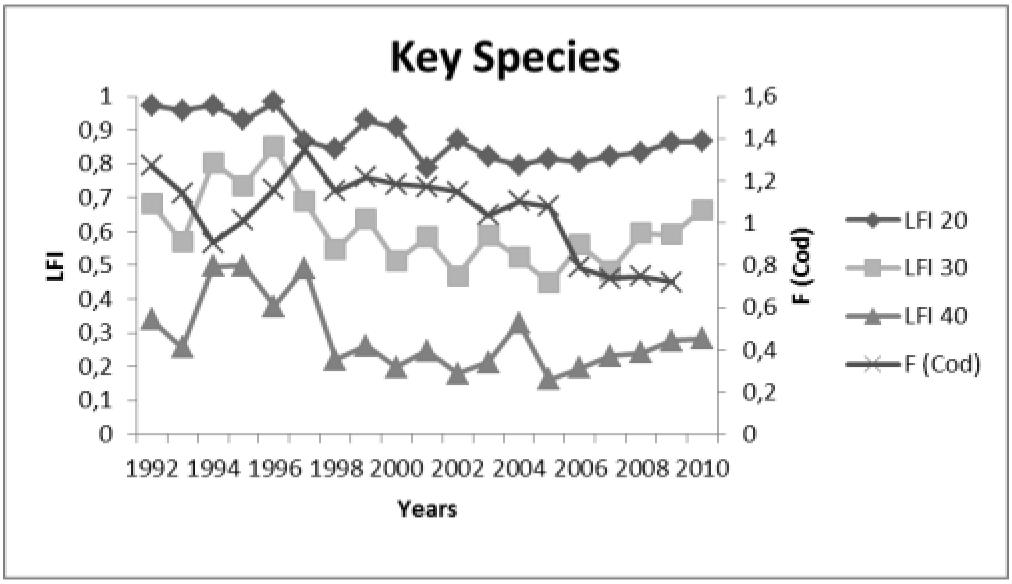

Figure 7.3.7. Results of the LFI calculated for the last year WGECO report (ICES, 2011).

7.3.2.4 References

ICES. 2011a. Report of the Working Group on the Ecosystem Effects of Fishing Activities (WGECO), 13–20 April, Copenhagen, Denmark. ICES CM 2011/ACOM:24. 166 pp.

ICES. 2011b: Data from the trawl survey database: http://datras.ices.dk/Home/Default.aspx.

SAS Institute Inc. 2009. JMP® 8 Discovering JMP. Cary, NC: SAS Institute Inc.

8 ToRf) Cumulative impacts from multiple pressures

ToR f. Cumulative impacts from multiple pressures

f. Carry out and report on a review of the state-of-the-art in understanding the combined effects of multiple ecosystem pressures, including advice for future research.

At many scales marine organisms, populations and ecosystems are subject to pressures resulting from wide ranges of human activities; from fishing and aquaculture to mining and wind-farms, and the disposal of a diversity of wastes. Besides these direct pressures, the marine environment is driven by climate variability. Therefore it is likely to be significantly affected by climate change, which generates a number of additional pressures. Water temperature range and variability change, but also dissolved oxygen and other nutrients, pH, as well as stratification and circulation. The ecosystem approach to fisheries implies to take account, not just of the full consequences of fishing on the ecosystem, but also the reverse: assess and manage the consequences of the ecosystem status and dynamics on fisheries. The latter is understandably often requested by fishers. The challenge here is that multiple pressures do not necessarily have simple, understandable, independent impacts. On the contrary, ecosystem dynamics are non-linear and one pressure might modify the response of an ecosystem component, or function, to another pressure –including fishing. It seems that little has been done so far to investigate how multiple pressures on the marine ecosystem interact. Below we first provide a few definitions useful in the context of multiple pressures. Second, we report evidence that interactions between pressures might be widespread, emphasizing the need to investigate these interactions. In section N.3, we report a review of the current understanding of the combined effects of multiple pressures: we recall the main pressures affecting the marine environment and list the potentially high number of linkages with ecosystem components, illustrating the wide scope for multiple pressures. We then report about methods developed to map multiple pressures, and present a limited number of studies that have examined the joined effects of two pressures (one of which was fishing). In Section 8.4 we outline some management issues generated by multiple pressures. In section 8.5 we suggest some avenues for further research in this area.

8.1 Definitions

We use the terms drivers, pressures, impacts, according to the DPSIR framework (Driving Forces-Pressures-State-Impacts-Responses). **Drivers**, or driving forces, are the environmental, socio-economic and socio-cultural forces driving human activities, which increase or mitigate pressures on the environment. We define **pressures** as "the mechanism through which a human activity has an effect on any part of the ecosystem". Pressures can be physical (e.g. abrasion), chemical (e.g. introduction of synthetic components) or biological (e.g. introduction of microbial pathogens) and the same pressure can be caused by a number of different activities. For example, both aggregate extraction and navigational dredging cause abrasion, a physical pressure that can affect a number of different ecosystem characteristics. **State**, or state of the environment, is the condition of the environment. **Impacts** are the effects of environmental change. **Responses** refers to the responses by society to the environmental situation. In the following climate change is considered as a driver of marine ecosystems; climate change is likely to interact with many pressures on the marine environment and modify their impacts. However, for the sake of brevity it will be included as a pressure whenever cumulative impacts of combined pressures are mentioned. We distinguish here climate change from climate variability – climate change only is considered to generate pressures – populations and communities are always living within an environmental envelope, of which variations might interact with human-induced pressures, but which are not included among the cumulative impacts considered in this section.

Cumulative impacts are considered here to be the impacts of two or more pressures acting simultaneously on the ecosystem. Such impacts may be additive, synergistic or antagonistic. **Additive effects** occur when the impact of one pressure is simply added to that of another pressure at a given scale, hence the impact of one pressure is unaffected by the level of the other pressure at that scale. The scale on which impacts are additive could be e.g. mortality, growth rate or recruitment success. Additivity at the given scale does not mean that impacts are also additive when examining transformations or functions of this scale (e.g. weight at age, number-at-age in the population). **Synergistic effects** occur when the impact of one pressure is increased by an increasing level of another pressure at all scales. For example, the growth induced by an increase in food abundance depends on the ambient temperature, large increases being prevalent at high temperatures and small increases prevalent at low temperatures. **Antagonistic effects** occur when the impact of one pressure is decreased by an increasing level of another pressure at all scales. For example, the mortality induced by an increase in predator abundance depends on the amount of alternative food available, large increases being prevalent when availability of alternative food is low and small increases prevalent when availability of alternative food is high. **Dominant effects** are a particular case of either type, where the impact of one pressure prevails over the consequences of other pressures.

The majority of investigations reported in the literature have focused on additive effects since these effects are reasonably straightforward to estimate and model. However, if synergistic or antagonistic impacts exist, they can potentially change the predictions based on the assumption of additivity entirely. The focus of this review is therefore mainly on synergistic and antagonistic impacts of multiple pressures and/or drivers.

8.2 Evidence of synergies and antagonisms

Experimental studies examining non-additive effects of multiple stressors are relatively recent; most have been published over the last decade. Their findings are summarized in two meta-analyses (Darling and Côté, 2008, Crain *et al.*, 2008). Darling and Côté (2008) examined interactions between pairs of stressors increasing animal mortality in freshwater, marine and terrestrial communities in 112 experiments, whereas Crain *et al.* (2008) summarized 171 studies from the marine environment using a broader range of stressor types. Both studies conclude that all types of interactions can occur and none seems to dominate: additive, antagonistic and synergistic interactions were all found in commensurate proportions in both studies. Moreover, Crain *et al.* (2008) highlighted that the addition of a third stressor changed interaction effects significantly in two-thirds of cases, and doubled the number of synergistic interactions.

This implies that the simplifying assumption of additive effects cannot generally be taken for granted, and that stressor interactions should be given more attention in research, impact assessments, and management.

8.3 State-of-the-art in understanding the combined effects of multiple pressures in the marine environment

8.3.1 Combining pressures

The basis for our evaluation of the occurrence and relevance of cumulative pressures was the so-called linkage framework developed in the FP7 project ODEMM (Options for Delivering Ecosystem-based Marine Management) (Koss *et al*., 2011). This linkage framework was intended to find all relevant sector-pressure combinations that may compromise the MSFD objective of achieving Good Environmental Status and should be applicable for all EU marine regions. To that end all the main sectors operating in the marine environment were selected, the pressures and ecosystem components as put forward by the MSFD were listed, and potential linkages (i.e. sector-pressure-ecosystem component) were identified. All linkages are one-way (sector/activities can cause a particular pressure). The linkages do not carry any weighting in terms of intensity, extent or frequency of each pressure relative to the sector/activity. Whenever a particular ecosystem component is affected by more than one sector-pressure combination there is scope for cumulative effects.

8.3.1.1 Sectors

The list of sectors (Annex 1) was generated and agreed through consultation within the ODEMM partnership (36 experts from 20 partners across Europe). It is: (i) inclusive of any sector that has had (historical), continues to (current), or is forecast (in the next 20 years) to exert pressure(s) that affect ecological characteristic(s) in any of the European regional seas; and (ii) broad in categorization of sectors so that ultimately this can relate back to socio-economic issues at national, international and/or regional levels. Sectors were identified with consideration of previous regional marine ecosystem assessment attempts including those undertaken in MEECE (Marine Ecosystem Evolution in a Changing Environment, www.meece.eu) and by OSPAR (2010).

Sectors are disaggregated into relevant activities because not all activities have the same pressures and it is thus possible to identify management options specific to individual activities rather than whole sectors. For example, abrasion caused by shipping is only associated with mooring activities while in port. The list of activities per sector was originally derived from the Marine Conservation Handbook (Eno, 1991) as amended by Cooke and McMath (2001), but this has been revised against other more recent lists.

8.3.1.2 Pressures

A total of 106 activities were attributed to 19 Sectors, each of which can contribute one or more human pressures to the marine ecosystem. The Marine Strategy Framework Directive (MSFD) lists 18 pressures (European Union, 2008); however this list has been expanded to 25 pressures for the ODEMM linkages (Table 8.1.1). The additional seven pressures (numbers 19 to 25) are considered as current or emergent threats to ecological characteristics.

Table 8.3.2.1.1. List of human pressures associated with sectors operating in Europe's regional seas.

Pressure Code	Pressure Name	Pressure Definition	Listed in the MSFD
1.	Smothering	Man-made structures or disposal of materials to the seafloor.	Yes
2.	Substrate loss	Sealing by permanent construction, e.g. Coastal defences, wind turbines.	Yes
3.	Changes in siltation	Suspended sediments in the water column from run-off, dredging, etc.	Yes
4.	Abrasion	Interaction of human activities with the seafloor and with seabed fauna/flora.	Yes
5.	Selective extraction of non-living resources	Sand & gravel extraction, exploration of subsoil, maerl extraction.	Yes
6.	Underwater noise	Underwater noise created from shipping, acoustic surveys, etc.	Yes
7.	Marine litter	Marine litter originates from numerous sources and consists of different materials including: plastics, metal, glass, rubber, wood and cloth.	Yes
8.	Thermal regime change	Change in thermal regulation due to climate change, or more locally due to outfalls.	Yes
9.	Salinity regime change	Regional salinity change due to climate change, or locally due to constructions affecting waterflow.	Yes
10.	Introduction of synthetic compounds	Introduction of pesticides, antifoulants, and pharmaceuticals into marine waters.	Yes
11.	Introduction of non-synthetic compounds	Introduction of heavy metals and hydrocarbons into marine waters.	Yes
12.	Introduction of radionuclides	Introduction of radionuclides into marine waters.	Yes
13.	Introduction of other substances	Introduction of solids, liquids or gases not covered by 10–12 or 14–15.	Yes
14.	Nitrogen and Phosphorus enrichment	Input of fertilisers, and other Nitrogen & Phosphorous rich substances.	Yes
15.	Input of organic matter	Organic enrichment e.g. from industrial and sewage effluent into rivers and coastal areas, from aquaculture.	Yes
16.	Introduction of microbial pathogens	Introduction of microbial pathogens into marine waters.	Yes
17.	Introduction of non-indigenous species and translocations	Introduction of non-indigenous species and translocations whether by sector activities or natural movement due to climate change.	Yes
18.	Selective extraction of species	Extraction of all species, including incidental non-target catch, e.g. by commercial fishing, recreational angling and collecting/harvesting.	Yes
19.	Death or injury by collision	Death or injury of marine fauna due to impact with sector activities, e.g. marine mammals with ships.	No

Pressure Code	Pressure Name	Pressure Definition	Listed in the MSFD
20.	Barrier to species movement	Stopping the natural movement of marine fauna due to barrages, causeways, wind turbines, and other man-made structures.	No
21.	Emergence regime change	Changes to natural regimes, e.g. widespread sea level rise due to climate change or locally due to barrages or other structures.	No
22.	Waterflow rate changes	Widespread change in currents due to climate change or local changes due to barrages and other man-made structures.	No
23.	pH changes	Widespread pH change due to climate change or localized affects, e.g. run-off from land-based industry.	No
24.	Electromagnetic changes	Electromagnetic energy emitted from electrical sources, e.g. underwater cables.	No
25.	Change in wave exposure	Regional wave change due to climate change, or localized due to installation of coastal structures.	No

8.3.1.3 Ecological characteristics

The list of 17 Ecological characteristics was derived from Table 1 (Annex III) of the MSFD and includes physical and chemical features, habitat types, biological and other (e.g. chemicals) features (Table 2). The pressure categories are described in Table 1 (Sectors-Human Pressures) above.

Table 8.3.1.3.1. List of ecological characteristics described in Annex III of the Marine Strategy Framework Directive.

Number	Ecological Characteristic
1.	Topography/Bathymetry
2.	Temperature
3.	Salinity
4.	Nutrients&Oxygen
5.	pH, pCO2
6.	Predominant Habitat Type
7.	Listed Habitat Types (e.g. N2K, SAC)
8.	Habitat Types Meriting Special Reference
9.	Plankton
10.	Bottom fauna and flora
11.	Fish
12.	Marine mammals & Reptiles
13.	Seabirds
14.	Species listed under Community Legislation or Conventions (e.g. Habitats Directive)
15.	Non-indigenous/exotic spp.
16.	Chemicals
17.	Other notable features

8.3.1.4 Linkages

All linkages are one-way (a pressure can affect an ecological characteristic). Linkages were evaluated using a combination of expert judgement and published literature and represent a direct link between a specific pressure and the characteristic. For example, indicating that the pressure, smothering, interacts with a particular type of habitat. The identified linkages do not infer an impact of the pressure on the ecological characteristic, merely that there can be an interaction (i.e. occurring at the same time in the same place).

The number of linkages is high, as illustrated for the example of Vertebrates in Table 8.3.1.4.1. For example, eight types of human activities are likely to generate selective extraction of species of either fish, mammal, or bird group. The last row shows the number of pressures potentially affecting each Vertebrate category: it ranges from ten for offshore seabirds to all 21 for demersal fish. Multiple pressures seem to be potentially the rule rather than the exception; the scope for multiple impacts is wide.

Table 8.3.1.4.1. Linkages between pressures (rows, from Table 8.3.2.1.1) and a subsample of ecosystem components (columns). The number in each cell is the number of human activities likely to generate that type of pressure on that component.

Pressures	Fish (deep-sea)	Fish (demersal)	Fish (pelagic)	Marine mammals	Seabirds – inshore	Seabirds – offshore	Grand Total
Barrier_to_species_movement		4	4	4	4	4	20
Change_in_wave_exposure		4	4	4	4		16
Changes_in_siltation	24	24	24	24	24	24	144
Death_or_injury_by_collision		12	12	12	12	12	60
Electromagnetic_changes	2	2		2			6
Emergence_regime_change		3	3	3	3		12
Input_of_organic_matter	11	11	11				33
Introduction_of_microbial_pathogens		12	12	12	12	12	60
Introduction_of_non_indigenous_species		9	9				18
Introduction_of_Non_synthetic_compounds	18	18	18	18	18	18	108
Introduction_of_Radionuclides	5	5	5	5			20
Introduction_of_Synthetic_compounds	23	23	23	23	23	23	138
Marine_Litter	14	14	14	14	14	14	84
Nitrogen_and_Phosphorus_enrichment	6	6	6				18
pH_changes		7	7	7			21
Salinity_regime_changes		8	8	8			24
Selective_extraction_of_species	8	8	8	8	8	8	48
Substrate_Loss	14	14	14	14	14	14	84
Thermal_regime_changes		6	6	6			18
Underwater_noise	21	21	21	21			84
Water_flow_rate_changes		14	14	14	14	14	70
Grand Total	146	225	223	199	150	143	1086
Number of pressures	11	21	20	18	12	10	11

8.3.2 Mapping multiple pressures

Human activities exert pressures on ecosystems on different scales; from local scales for extraction or contamination through regional scales for exploitation of marine populations to global scale for climate change. On these scales not all pressures are continuous and uniform. Therefore for any particular ecosystem, evaluating multiple impacts starts with identifying the places and times where human activities actually generate multiple pressures. Indeed the first attempts at evaluating "cumulative impacts" consist in mapping several types of pressures and finding a common currency and operator for combining them. At a global scale, Halpern *et al.* (2008b) combined maps of 17 pressures from major activities (including fishing, pollution, climate change, shipping and invasive species) with maps of 20 habitat types. "Impact" was scored for 1 km² cells as the sum of pressure intensity × habitat occurrence weighted by an expert-based index of sensitivity of each habitat to each pressure. Summation implies the assumption that interactions between pressures are negligible. The study concluded that over 40% of the global ocean is submitted to multiple pressures. Applying the method on a more local scale, Ban *el al*. (2010) found that all of Canada's Pacific waters are affected by multiple activities. A risk-analysis framework was developed and applied to UK marine waters to combine the pressures exerted by 8 human activities on 20 landscape types (Stelzenmüller *et al.*, 2010). Pressures were ranked by relative importance and risk was quantified as a weighted sum of standardized pressure intensity; weights being assigned to each rank by a variety of scenarios (equal, linear, …). The outcomes are sensitive to changes in the ranks, and in the standardization and combination parameters; however, in most configurations, a significant part of UK waters were found to be subject to high levels of risk. Another cumulative risk analysis for combined pollutants in Danish soils (Lahr *et al.*, 2010) also outlined the sensitivity of the results to the assumptions made when combining pollutants effects (dominance, addition, synergy or antagonism). Even when combining only two pressures, benthic fishing and aggregate extraction, "cumulative impact assessment" consisted in calculating recovery rates based on a range of scenarios combining the known impacts of each pressure taken separately: from antagonistic through additive or dominant to synergistic (Foden *et al.*, 2010). The results again are highly dependent on the combination scenario.

8.3.3 Understanding the combined effects of pressures: existing knowledge

8.3.3.1 Fishing predatory, cannibalistic and prey species

In numerous ecosystems, fisheries targeting both predatory, cannibalistic and prey fish present cumulative impacts, as the impacts of fishing pressure on biomass depend both on the stock size and on natural mortality. Natural mortality is impacted by the fishing mortality on predatory fish (including cannibalistic species) as increasing fishing mortality decreases predator abundance, thus prey natural mortality if all other aspects remain constant. Further, fishing pressure on alternative prey species affect natural mortality as an increasing pressure leads to a decrease in the abundance of alternative food and hence an increase in natural mortality assuming that the predator shows at least some degree of a compensatory response. Depending on the parameterization of the model, impacts of pressures can be additive (BALMAR (Lindegren *et al.*, 2009), multispecies production models) or non-additive (EwE, Multispecies assessment models). Unfortunately indirect fishing impacts are often confounded or counteracted by direct removals, precluding conclusions about the nature of the interaction (e.g. Cox *et al.*, 2002; Friedlander and deMartini, 2002; Rochet *et al.*, 2010).

A recent example of impacts of fishing on interacting stocks is given by WKMULTBAL (2012). The working group investigated the combined effects of fishing for cod, herring and sprat in the Baltic Sea using a suite of dynamic models of these three species. The non-additive model shows clear non-linear effects when compared to the additive model (Figures 8.3.3.1.1 and 8.3.3.1.2), as yield in the additive model is linear in fishing mortality with no change in slope of the relationship as other pressures are varied. Increasing fishing mortality for the predator increased the yield of the two prey species considerably even after passing the fishing mortality leading to maximum yield of cod (Figure 8.3.3.1.2). There are numerous examples of similar exercises for other areas. The comparison presents a clear example of the danger of using models to infer the presence or absence of non-additive effects, as this is basically given by the specification of the model. The same applies to many empirical analyses which specify a given statistical model.

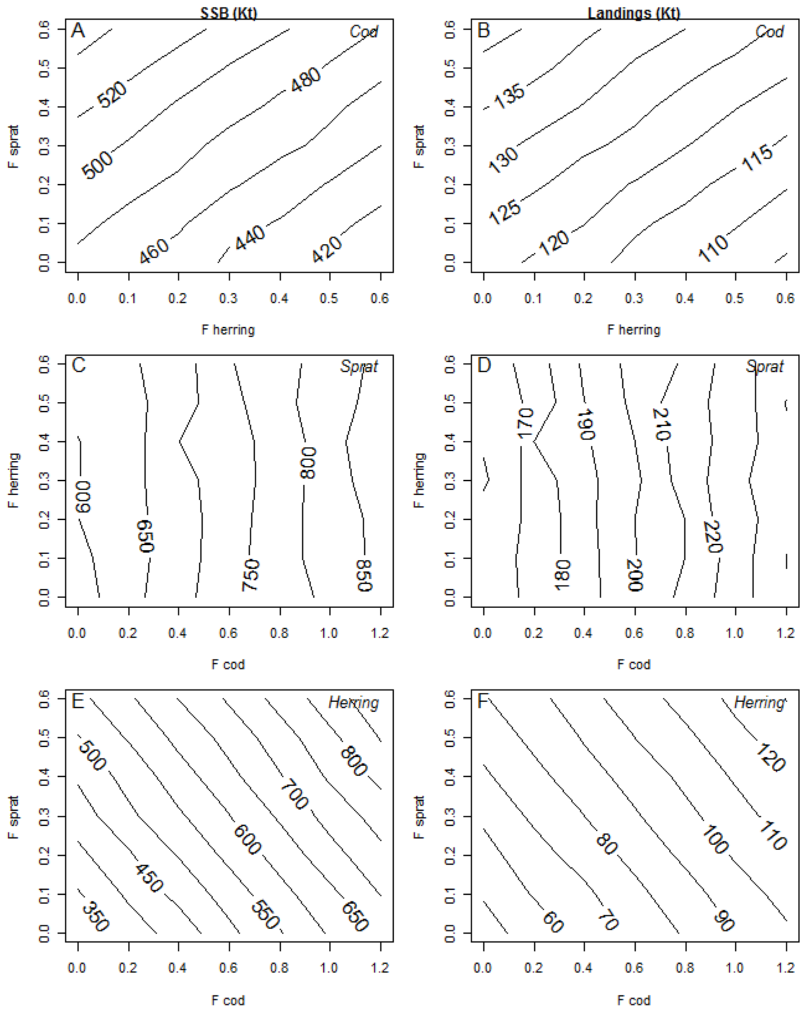

Figure 8.3.3.1.1. Average forecasted spawning–stock biomasses (SSB) and landings of cod (A, B), sprat (C–D) and herring (E–F) from the BALMAR model. Exploited at the recommended reference F for each species separately (F_{cod}=0.3; F_{sprat}=0.32; $F_{herring}$=0.16), while varying the fishing mortality for the two other species under a simulated period of ten years initiated in 2010. Source: WKMULTBAL, 2012.

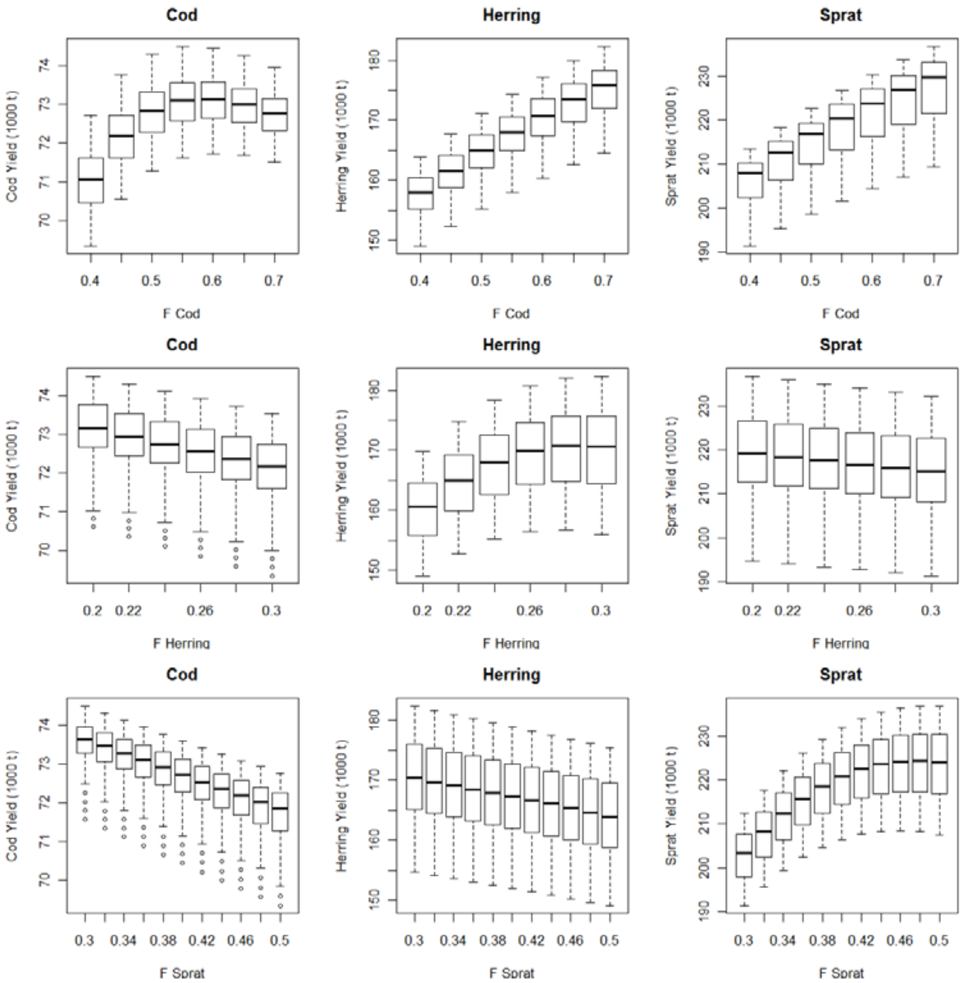

Figure 8.3.3.1.2. Box plots of scenario results of Yield at equilibrium from an SMS model (Lewy and Vinther, 2004) of the Baltic for combinations of F on the three species. Stochastic recruitment. Source: WKMULTBAL, 2012.

8.3.3.2 Fishing and climate

Many studies in the past have aimed at disentangling climate from fishing effects on populations; since fishing effects are deemed more manageable. The understandable intention was to provide proof of fishing impacts to justify management measures and improve acceptability compliance by stakeholders. These studies relied on the implicit assumption of additive or dominant effects. However, there is increasing evidence of interactions between fishing and climate impacts, reviewed by Planque *et al.* (2010). The depletion of large and old individuals induced by fishing affects population capacity to buffer environmental variability, as does the selection of population subunits within meta-populations. Fishing has also been shown to increase the sensitivity of stock distributions to climatic variation (Rindorf and Lewy, 2006). At the ecosystem level, fishing by modifying the relative proportions of species and promoting increasing turnover rates might alter resilience to perturbations (Planque *et al.*, 2010). Conversely, climate change is triggering adaptations and changes in the phenology of marine populations and communities. Seasonal changes in temperature, acidification, and daylength have or are expected to have consequences on the physiology and timing of key life-history events (Hollowed *et al.*, 2011) that are going to affect the productivity and population dynamics; hence the population and community responses to fishing.

8.3.3.2.1 Climate and fish stocks

Numerous studies have investigated the relationship between changes in productivity caused by e.g. climate and sustainable fishing pressure of individual stocks. Climate often affects the stock at multiple levels with both synergistic and antagonistic effects possible, resulting in changes to the impact of fishing (Kell *et al.*, 2005; Lindegren *et al.*, 2009). Other studies have shown changes in sustainable fishing levels as stock productivity changed (Mohn and Chouinard, 2007) and effects of changes in recruitment (A'mar *et al.*, 2009a, b; Brunel *et al.*, 2010).

8.3.3.2.2 Climate and the fish community

How fishing and climate could affect marine communities has been investigated under the assumption that fishing impacts should propagate top–down with mortality of predator releasing prey from mortality, increasing mortality of the prey resource in turn, thus generating negative correlation patterns between successive trophic levels. By contrast, climate effects would propagate bottom–up from the primary production level towards higher trophic levels (Cury *et al.*, 2000). Much research has sought to identify the dominant effect (Frank *et al.*, 2008), once again ignoring potential interactions. In a comparative study across the North Atlantic, top–down control has been found to dominate in northern areas while bottom–up control governed the predator–prey dynamics in southern areas (Frank *et al.*, 2008). Temperature, which influences the demographic rates of the component species, may provide an explanation for the resilience to overfishing effects in the southern areas; illustrating an interaction between climate and fishing impact propagation. Recent attempts to analyse time-series of fishing pressure, environmental drivers and community metrics have shown the importance of both drivers across Northwest Atlantic fish communities (e.g. Shackell *et al.*, 2012). Another comparative study has shown that in several Northeast Atlantic and Mediterranean temperate shelf fish communities, the expected trends induced by changes fishing and climate are in the same direction; the impacts are therefore confounded, making it difficult to quantify the interaction (Rochet *et al.*, 2010).

8.3.3.2.3 Acidification

CO_2 from fossil fuel emissions entering the ocean generates water acidification. Global surface pH has already decreased compared to pre-industrial times, and might decrease further over the next decades. Ocean acidification hinders growth of calcium carbonate shells of many marine plants and animals, including harvested shellfish species. In 2007, mollusc fisheries contributed 19% or $748 million to the total exvessel value of US fisheries (Cooley and Doney, 2009). These authors estimated the potential losses due to acidification based on laboratory experiments showing damage to individual organisms combined with 50-year projections of CO_2 concentrations. As a first approximation, the authors assumed that reduced calcification rates at a given CO_2 concentration would proportionally decrease harvests. With this approximation, an estimated 10–25% decrease in calcification rate would translate to $75–187 million losses. This calculation assumes that acidification has an additive effect that essentially reduces the productivity of shellfish stocks. One of the management implications is that fishery management plans could be updated to include the effect of acidification. The authors acknowledged several caveats to these calculations but felt that they provide a first-order approximation of the potential effects of ocean acidification on shellfish harvests. A similar first-order approximation using a dynamic bioclimatic envelope model that incorporates hypotheses on the effects of changes in ocean biogeochemistry and phytoplankton community structure on fish distribution and productivity in the Northeast Atlantic suggests that the resulting reduced growth performance and range shift might significantly lower the long-term (2050) expected catch potentials (Cheung *et al.*, 2011). However, the rate of ocean acidification is not yet well predicted, and the effects of pH changes on biological processes and their rates need to be further investigated, especially in the context of multiple environmental variables changing at the same time (Denman *et al.*, 2011).

8.3.3.2.4 Ecosystem level analyses of climate impact

Bayesian Networks (BN) have become increasingly popular methods to model uncertain and complex domains such as ecosystems, and in environmental management. The approach is being explored for use in integrated ecosystem assessment in the North Sea by ICES WGINOSE (Working Group on Integrated Assessments of the North Sea). The information presented here draws on work by Vanessa Steltzenmuller and Ulrich Callies for WGINOSE.

Bayesian networks, are also called Belief networks, Bayesian belief networks, Bayes nets or probabilistic networks, and are one branch of Bayesian modelling (i.e. hierarchical simulation based modelling). The basic idea is of conditional dependence between variables and the updating of knowledge based on Bayes' theorem: $p(B|A) = p(A|B) \times p(B) / p(A)$. So they are considered very useful for evaluating combinations of pressures that may interact, including for developing our understanding of synergistic and antagonistic interactions (Lemmer and Gossink, 2004). An example application in the context of fisheries interactions is presented in Stelzenmuller *et al*. (2011). This analysis looked at the interactions of environmental factors (bottom temperature, salinity and depth) with fishing effort by métier, to predict the consequences for plaice vulnerability to fishing. The BN is presented in Figure 8.3.3.2.4.1.

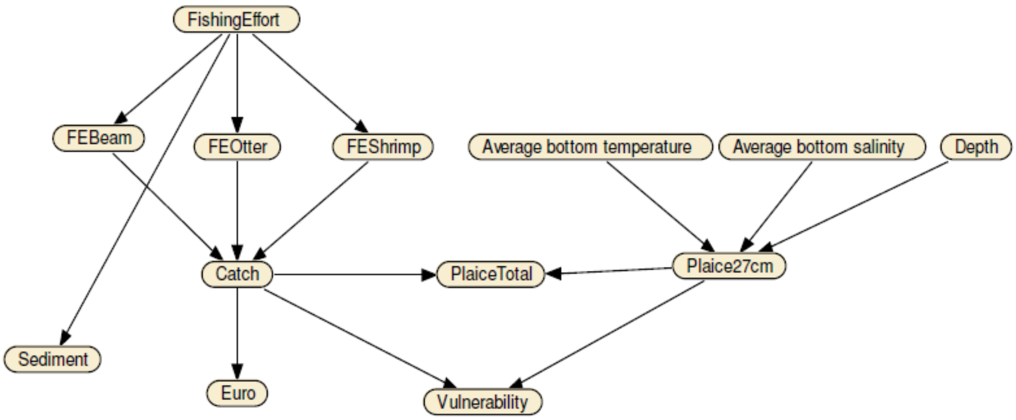

Figure 8.3.3.2.4.1. Conceptual model showing the key variables used to predict the overall level of vulnerability of plaice *Pleuronectes platessa* to fishing as a function of total catch and catch per unit of effort (cpue) of plaice ≥27 cm. The resulting model was then used in combination with a fishing displacement simulation to examine the effects of exclusions for wind farms.

8.3.3.3 Alien species

Alien species invasion may alter the structure and functioning of marine habitats and ecosystems and their response to fishing-induced disturbance. In the Venice lagoon, the introduction of Manila clam (*Ruditapes philippinarum*) for aquaculture purposes gave rise to a change in the structure of benthic community with a massive spread of this species (Pranovi *et al*., 2006). The combination of this process with intense mechanical clam harvesting fostered a shift from a herbivore–detritivore dominated community to a filter-feeders dominated one (Pranovi *et al*., 2008). This shift determined variations in the secondary production and induced modifications in the type of ecosystem control possibly inducing changes on the dynamic stability of this ecosystem.

Overexploitation can also increase the vulnerability of ecosystems to alien species invasion. Daskalov *et al*. (2007) investigated regime shifts in the Black Sea, once described as healthy and dominated by various marine predators, which by the late 20th century experienced anthropogenic impacts such as heavy fishing, agricultural eutrophication, and invasion by alien species. Two major shifts were detected, the first related to a depletion of marine predators (owing to overexploitation) and the second to an outburst of the alien comb jelly *Mnemiopsis leidyi.* The authors hypothesized that both shifts were facilitated by intense fishing resulting in system-wide trophic cascades. Indeed *M. leidyi* population expanded in 1990 when decreased zooplanktivory by overfished stocks increased surplus zooplankton production, which was used by the burgeoning *M. leidyi* population. However, there have been other suggestions on the cause of the comb jelly increase (Shiganova and Bulgakova, 2000).

8.3.3.4 Pollution effects on fish stocks

Pacific herring (*Clupea pallasi*) in Prince William Sound (PWS) were affected by two major events in the 1990s: the Exxon Valdez oil spill in 1989 and a 75% collapse in the adult population in 1993 (Carls *et al*., 2002). In this review paper, the authors compared and reinterpreted published data from industry and government sources. Based on laboratory effects thresholds of 0.4–0.7 $mg \cdot l^{-1}$ total polynuclear aromatic hydrocarbons, and site-specific estimates of exposure, they concluded that 25–32% of herring embryos were damaged in PWS in 1989. Significant effects extended beyond those predicted by visual observation of oiling and by toxicity information available in 1989. Oil-induced mortality probably reduced recruitment of the 1989 year class into the fishery, but this effect was impossible to quantify because recruitment was generally low in other Alaskan herring stocks. Significant adult mortality was not observed in 1989; biomass remained high through 1992 but declined precipitously in winter 1992–1993. The collapse was likely caused by high population size, disease, and suboptimal nutrition, but indirect links to the spill cannot be ruled out. In this example, mortality due to oil exposure was an additive source of mortality to herring embryos. However, it was difficult to determine whether this additional source of mortality was damped or amplified by processes between the larval stage and recruitment to the herring fishery three years later.

8.4 Management

Management of a fishery undergoing the influence of other pressures or drivers could rely on model-based predicted outcomes of management actions under various scenarios for the other pressure(s). Some tools are already available for fishery management in a changing environment. Harvest control rules based on fishing mortality estimates are generally fairly robust to changes in the environment, as long as these changes remain within an acceptable range (Kell *et al.*, 2005; A'mar *et al*., 2009; Brunel *et al*., 2010). In certain settings however, more adaptive control rules might work better in a changing environment (Ianelli *et al*., 2011). Stock reference points tracking the environmental conditions do not necessarily perform better than conventional reference points (A'mar *et al*., 2009). By contrast, a more comprehensive approach with an ecosystem model (Atlantis) has shown that the performance of a given management method (e.g. individual fishing quotas) might be significantly affected when the impact of ocean acidification is accounted for (Kaplan *et al*., 2010).

These developments however face daunting challenges owing to high uncertainties in climate change predictions and ecosystem models. Thus far, there has been moderate success in modelling environment–plankton relationships, to say nothing about environment–fish recruitment correlations. Because ecosystem response are both complex and usually non-linear, predicting consequences of climate change for low- to high-trophic level species and communities remains a challenge, and uncertainties in ecosystem models, from model structural to parameter uncertainty, could confound climate impacts (Plagányi *et al.*, 2011). For testing management procedures the use of simplified surrogates for climate change effects might be advisable; especially as on the other end, more components, including economic and socio-cultural factors which are key concerns under changing climate, might have to be included in an operating model (Plagányi *et al.*, 2011).

Climate change, and more generally multiple pressures, create a context of increasing variability and uncertainty, where flexible management policies are likely to be increasingly required. It may not be necessary to predict changes (such as climate change) nor all consequences of cumulative impacts. Rather, management strategies that can be readily adapted to changing conditions or new knowledge are going to be increasingly needed. Dealing with uncertainty will remain a key challenge for management, as it is today. Plagányi *et al.* (2011) identify two critical needs to address this challenge: increasing the flux of data and knowledge by reducing the cost per observation and analysis, and simplifying and streamlining conflict resolution processes.

The two most difficult issues when managing multiple impacts are likely to appear when multiple activities are involved and/or when trade-offs must be made between conflicting objectives. Human activities operate over various spatial and temporal scales. Ocean zoning might be a useful way of identifying areas deserving coordinated management of several pressures (Halpern *et al*., 2008a). Once the scales, times and places where combined pressures happen and might generate cumulated impacts are identified, management actions need to be traced back to the various activities involved: multiple agencies operating at different scales with different methods and tools might need to be involved and cooperate, to trade-off impacts vs. costs and benefits of activities. This is not likely to be simple.

8.5 Summary and research needs

Potential interferences between multiple pressures are likely to be widespread; and the first attempts at mapping multiple pressures suggest that this potential overlap is realized in many parts of the marine environment. The scope for cumulative impacts is therefore wide. Moreover, both the few experimental studies available, and some worked examples reported above, suggest that, more often than not, interactions between multiple pressures are not simple; with both synergistic and antagonistic effects happening. But unfortunately, most available evidence is qualitative or blurred by confounding factors; whereas we can consider there is convincing evidence that cumulative impacts do occur, we keep short of quantitative estimates of even their magnitude and importance. Thus, there is a pressing need to identify in which cases synergies between pressures are likely to affect marine ecosystem more than single-pressure studies would suggest, for these might require fast, coordinated mitigation measures. It would also be interesting to be aware of antagonisms, which may in some cases have concealed the actual impact of extant pressures. Should some of the extant pressures be decreased, the remaining pressures which are involved in antagonistic effects may have unexpectedly large effects. As we know that synergistic and antagonistic effects may occur and may have serious implications, simply assuming that these effects are absent is not precautionary. A first step towards identifying the most likely, or most threatening, interactions would be an expert-based analysis of the linkage table shown in Section 8.3.1.4. This table is so far exempt of any ranking between pressures; obviously the interactions between the pressures likely to exert the strongest impacts need to be investigated in priority.

Unfortunately, the study of synergistic and antagonistic pressures is complicated by the often very large variation in natural ecosystems. These make it difficult to identify statistically significant interactions between pressures from observational studies, and often leads to cases where simple additive models conform equally well to data as more complicated non-additive models. Hence, traditional model selection does not necessarily result in models which have predictive reliability outside the range of pressures recorded historically. Further, when quantitative models with significant non-additive terms are constructed, computational time often becomes limiting as complexity increases.

An important part of cumulative impacts research will require multidisciplinary work, up to social and economic sciences. Murawski (2011) lists a number of research tracks towards progress in integrating climate change science and fisheries, beyond improving the "Earth System Models" (ESMs) themselves, their scope and comparability. The development of integrated models will be necessary to better frame research needs and support management; as well as for complex statistical models, this might require increased computational power. Understanding multiplicative factors and interactions cannot be done sequentially; a holistic approach to these analytical issues is required. A better understanding of the synergistic effects of climate forcing and socio-economic forces is required if we are to understand the viability of adaptation strategies.

But this will not suffice. Interactions are by definition surprises, and therefore first need to be investigated empirically; even if biological knowledge and ecological theory can help forecast and understand the underlying mechanisms. There is a need for field studies of the most important interactions at various scales. Integrated observation systems are going to be increasingly needed, with a wide variety of physical, biological, ecological and human observations collected on potentially large-scales.

We can take examples from e.g. eco-toxicological studies, where the assumptions for combining single toxic compounds can more easily be based on laboratory experiments where cumulative effects are empirically tested (e.g. Jonker *et al.*, 2005), and underpin an acceptable rationale for the obtained results. Denman *et al.* (2011) list a number of recommendations for future experimental studies of the effects of ocean acidification on marine organisms, especially the need of accounting for simultaneous changes in multiple stressors (in particular increasing temperature and decreasing dissolved oxygen). Experimental protocols should include behavioural and physiological dependencies on multiple variables that are expected to change with the climate: e.g. P_{CO_2}, dissolved oxygen, temperature, and micronutrients such as iron.

Unfortunately, conventional experimental approaches may not be applicable to all pressures of which cumulative impacts are to be expected. Some pressures such as those generated by climate change cannot be easily manipulated at a scale commensurate with the other pressure(s) of interest, or at a scale relevant to the management issues. When experimental systems can be created to examine a given interaction, it might be difficult to infer how the results apply to real systems. For example, whether experimental results about fishing-induced evolution (Conover and Munch, 2002) can be extrapolated to wild populations is subject to debate (Conover *et al*., 2005); it is likely that this kind of question would be even more difficult when multiple factors are manipulated; if ever they can be manipulated on commensurate scales. Here, active adaptive management *sensu* Walters (1986) may be required. This consists in designing management strategies with contrasted options, with the objective of learning from the outcomes. This approach should be particularly relevant to investigate cumulative impacts of multiple pressures.

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Annex 1. The list of sectors that contribute at least one pressure to at least one of Europe's regional sea ecosystems

Sector	Code	Description of Activities
Aquaculture	1	Finfish
		Macroalgae
		Predator Control
		Shellfish
Fishing	2	Benthic trawls (e.g. scallop dredging)
		Fishery wastes
		Netting (e.g. fixed nets)
		Pelagic trawls
		Potting/creeling
		Suction (hydraulic dredging)
Shipping	3	Litter and debris
		Mooring/beaching/launching
		Shipping
		Shipping wastes
		Ferrying people
Renewable Energy	4	Renewable (tide/wave/wind) power station
Non-renewable Energy (oil & gas)	5	Oil & Gas
		Power stations
		Thermal discharge (cooling water)
		Water resources (abstraction)
Non-renewable Energy (Nuclear)	6	Nuclear effluent discharge
		Nuclear power
		Thermal discharge (cooling water)
Telecommunications	7	Communication cables
Aggregates	8	Inorganic mine and particulate waste
		Maerl
		Rock/Minerals (coastal quarrying)
		Sand/gravel (aggregates)
Navigational Dredging	9	Capital dredging
		Maintenance dredging
		Removal of substrate
		Spoil dumping
Coastal Infrastructure	10	Artificial reefs
		Barrage
		Beach replenishment
		Communication cables
		Construction phase
		Culverting lagoons
		Dock/port facilities
		Groynes
		Land claim
		Marinas

Sector	Code	Description of Activities
		Oil & Gas platforms
		Removal of substrate
		Sea walls/breakwaters
		Urban
Land-based Industry	11	Industrial effluent discharge
		Industrial/urban emissions (air)
		Particulate waste
Agriculture	12	Agricultural wastes
		Coastal farming
		Coastal forestry
		Land/waterfront run-off
Tourism/Recreation	13	Angling
		Boating/Yachting
		Diving/Dive site
		Litter and debris
		Public beach
		Tourist Resort
		Water sports
Military	14	Military
Research	15	Animal Sanctuaries
		Archaeology
		Research
Desalination	16	Effluent discharge
		Water resources (abstraction)
Wastewater Treatment	17	Sewage discharge
		Thermal discharge
Carbon Sequestration	18	Exploration
		Construction
		Operational
Collecting/Harvesting	19	Bait digging
		Seaweed and saltmarsh vegetation harvesting
		Bird Eggs
		Shellfish hand collecting
		Peels
		Curios

9 ToRg) Trade-off in biodiversity conservation and sustainable use

Develop advice on approaches to handling trade-offs in biodiversity conservation, in particular across the three pillars of sustainability.

This ToR continues and extends WGECOs 2011 ToR E: Strategic Initiative on Biodiversity (ICES, 2011a). While many biodiversity initiatives concentrate on defining, measuring, and mapping biodiversity (e.g. ICES Working Group on Biodiversity Science; ICES, 2011b), WGECO examines the trade-off between biodiversity conservation and sustainable use. In brief, WGECO catalogued the objectives of marine conservation and sustainable use and defined the important terms (ICES, 2011a). WGECO examined the extent to which these objectives are aligned and the extent to which trade-offs can be mitigated by management measures. The guidance provided included (1) a framework for assessing biodiversity in marine ecosystems, (2) best practices for setting reference levels that reflect sustainable use, (3) how to assess when components of biodiversity are subject of serious or irreversible harm, (4) guidance on the use of expert judgment, and (5) best practices for setting reference points in changing conditions.

Here we extend our previous work across the three pillars of sustainability by examining trade-offs between biodiversity conservation and social and economic objectives. This section starts by reviewing the socio-economic objectives of conventions and acts that pertain to conservation and sustainable use of marine resources. We cite the biodiversity indicators and their proposed reference levels, and list a number of economic and social indicators that have been used in marine resource contexts. We review various approaches that have been used to handle trade-offs between biodiversity and socio-economic indicators and provide several examples.

9.1 Objectives of marine conservation and sustainable use of marine resources

WGECO (ICES 2011a) reviewed the high-level objectives of conventions and acts that pertain to the conservation and sustainable use of marine habitats, fauna, and flora. In this section we focus on the socio-economic aspects of these objectives.

9.1.1 Convention on Biological Diversity (CBD)

The CBD lists three overarching objectives, which are equal in importance: "The objectives of this Convention, to be pursued in accordance with its relevant provisions, are the conservation of biological diversity, the sustainable use of its components and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources, including by appropriate access to genetic resources and by appropriate transfer of relevant technologies, taking into account all rights over those resources and to technologies, and by appropriate funding." (http://www.cbd.int/).

The term conservation is never defined in the Convention, but it cannot be taken as protect in (near) pristine state, because the sustainable use objective is equal to the conservation one. Sustainable use *is* defined in the Convention. "Sustainable use means the use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations." Moreover, a later provision of the Convention calls on Parties to "Adopt measures relating to the use of biological resources to avoid or minimize adverse impacts on biological diversity"; this provision again legitimizes use of biological resources, and the adjective "adverse" is not accidental. It is acknowledged that biodiversity will be impacted through use. The goal is to minimize impacts that are adverse.

The third objective on equitable sharing of benefits differentiates the CBD objectives from the MSFD or CFP in Europe, or even the National Ocean Policy in the US or Oceans Act in Canada. It puts social well-being and viable livelihoods on an equal plane with economic value, profit, etc. The CBD has fully embraced economic valuation of biodiversity as a valid and useful concept and approach (notwithstanding the opposition to valuation by some Parties). In fact, Aichi Biodiversity Target 2 says "By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems." (http://www.cbd.int/sp/targets/). So economic value of use of biodiversity is relevant currency.

WGECO noted that the "equitable sharing of benefits" is taken very seriously. The social consequences of how much and how biodiversity is used are just as important as the economic consequences. For many of the biodiversity-rich parts of the world, a large fraction of the populations pursue subsistence lifestyles tied to uses of biodiversity. The CBD is clear that the goal should not be to convert such subsistence uses of biodiversity to market-based uses. Rather the goal is "equity" in the distributions of benefits from the uses.

9.1.2 Marine Strategy Framework Directive (MSFD)

The preamble of the MSFD Directive (EC 2008) states: "As a first step in the preparation of programmes of measures, Member States across a marine region or subregion should undertake an analysis of the features or characteristics of, and pressures and impacts on, their marine waters, identifying the predominant pressures and impacts on those waters, and an **economic and social analysis** of their use and of the cost of degradation of the marine environment." Annex IV, an indicative list of characteristics to be taken into account for setting environmental targets, suggests "Due consideration of social and economic concerns in the setting of targets." should be made. There is no mention of specific economic or social objectives that should be considered within the MSFD. The Commission Decision on MSFD makes no mention of economic or social considerations (EC, 2010a). However, the front page of the EU website on MSFD, states that the MSFD "aims to achieve good environmental status of the EU's marine waters by 2020 and to protect the resource base upon which marine-related economic and social activities depend."

http://ec.europa.eu/environment/water/marine/directive_en.htm

In a study commissioned by DEFRA in the UK, but considering the whole OSPAR Region, the outlines of such an "economic and social analysis" ESA were laid out: (http://randd.defra.gov.uk/Document.aspx?Document=me5103_9928_FRP.pdf). These extensive reports include the following elements:

- Consideration of the socio-economic effects of choosing targets;
- Requirements for Cost–benefit Analysis of any new measures;
- Implementation of measures that are cost-effective, and introducing economic incentives that support GES; and
- Exceptions for reasons of overriding public interest or where costs are disproportionate."

They also recommend that the approach to the ESA adopt the four steps proposed by Turner *et al*. (2009) for OSPAR, setting out a socio-economic decision-support system:

- To use the Driver-Pressure-State-Impact-Response framework, with the term 'impact' referring to changes in human welfare;
- To use the ecosystem services approach, so that changes to the environment are characterized in terms of their links to goods and services provided to humans or as inputs to other ecosystem services;
- To encourage adoption of IPCC scenarios for consistency and comparability of analysis;
- To use an extended Cost–benefit Analysis (CBA) framework, which can accommodate non-monetary information where necessary, and can adapt to undertake other styles of analysis (e.g. cost-effectiveness analysis and multi-criteria analysis), where appropriate and desired by Member States." CBA would include market and non-market costs and benefits. The social analysis was assumed to supplement the economic analysis by putting more emphasis on employment impacts and the distribution of economic impacts among different groups in society.

9.1.3 The Common Fisheries Policy (CFP)

The 2002 CFP states that "the Common Fisheries Policy shall ensure exploitation of living aquatic resources that provides **sustainable economic, environmental and social conditions**." The CFP is currently undergoing a review process for a planned new implementation in 2012. A Green Paper issued on this process for consultation (EC, 2009), goes on to say "No priority is set for these objectives" and it is not clear how this relates to economic and social conditions. There are no clear indicators that could provide more concrete guidance or help measure policy achievements.

The Green Paper then states "Economic and social sustainability require productive fish stocks and healthy marine ecosystems. The economic and social viability of fisheries can only result from restoring the productivity of fish stocks. There is, therefore, no conflict between ecological, economic and social objectives in the long term." This statement appears to imply that, in the opinion of the commission, there are no long-term trade-offs between ecosystem, economic and social objectives. They go on to indicate that in the short term, the rebuilding of fish stocks can result in economic and social impacts. They also observe that pursuit of social objectives, e.g. in employment, especially in the short term, can lead to a failure to rebuild stocks and jeopardize the economic future of fisheries.

This part of the Green paper concludes with the question: "How can the objectives regarding ecological, economic and social sustainability be defined in a clear, prioritized manner which gives guidance in the short term and ensures the long-term sustainability and viability of fisheries?" More specifically, and in the social domain: "Should the future CFP aim to sustain jobs in the fishing industry or should the aim be to create alternative jobs in coastal communities through the IMP and other EU policies?"

In a specific proposal, the Green Paper suggests differentiated management for the small-scale inshore fishing fleets. They say "There is a legitimate social objective in trying to protect the most fragile coastal communities from this trend" of reduced employment. This approach is also extended to small-scale fisheries in the wider world context.

The EC also published a Synthesis of the Consultation on the Reform of the Common Fisheries Policy (EC, 2010b). One key point to emerge from this consultation was that "It is generally agreed that ecological sustainability creates the basis for a viable fishing sector, with little long-term conflict between ecological, social and economic objectives." Job creation was identified as important, by Member States (MS) and industry. Food supply and food security were also identified as key objectives especially by MS. There appeared to be strong support for the protection of small-scale fishing communities, and the link to the local/regional community. Some proposed fishery-based protection (aka ring fencing) and a few suggested inclusion of recreational fisheries in the specific regime. Throughout the document, responses seemed to indicate a need to consider the social and economic implications of the CFP, without going into specifics. Other social objectives raised, included: attractiveness of the profession, recognition of the role of women and vulnerable groups in fisheries.

In both documents, economic factors are mentioned extensively, usually in terms of economic viability, efficiency and sustainability, but generally without specific objectives.

The EC also published a document entitled "The Social Dimension of the CFP reform" (http://ec.europa.eu/fisheries/reform/docs/social_dimension_en.pdf). This document highlights employment issues, including declining employment in the fishery sector, and decreasing "attractiveness" of such employment, mainly suggested to be due to low incomes. The document highlights a number of "social objectives" of the reformed CFP:

- reversing the decline in employment in the fisheries sector, particularly in catching;
- increasing the attractiveness of the fisheries sector and turning it into a source of high quality jobs;
- ensuring the viability of coastal communities by promoting economic growth and jobs;
- facilitating the transition to a sustainable fishing;
- unlock the potential of European aquaculture to expand and create new jobs in inland as well as in marine aquaculture.

Again, these focus on employment and on coastal communities. The document reiterates that "Achieving environmental sustainability as quickly as possible is a precondition for social sustainability".

In summary, the CFP reform explicitly focuses on social and economic aspects of policy. On the economic side, this addresses viability, efficiency and sustainability, but generally without specific objectives. On the social side, the emphasis in on employment and protection of coastal communities, although generally the latter is seen as involving improved employment.

9.2 Biodiversity indicators, targets, and biodiversity loss

In the context of the MSFD, WGECO has conducted extensive reviews of ecological indicators, including the biodiversity descriptors (ICES, 2011a). WGBIODIV (ICES, 2011b) has also given some consideration to indicator development. The MSFD, EC Decision Document (EC, 2010a) contains an extensive list of biodiversity indicators, some of which have been assigned reference levels.

A key paragraph in "OSPAR's MSFD Advice Manual on Biodiversity: Approaches to determining good environmental status, setting of environmental targets and selecting indicators for Marine Strategy Framework Directive descriptors 1, 2, 4 and 6", prepared by the OSPAR Intersessional Correspondence Group on the Coordination of Biodiversity Assessment and Monitoring (ICG-COBAM), reads:

> *"The TG1 report provided guidance on the interpretation of Descriptor 1, whereby the aim to have biodiversity 'in line with prevailing physiographic, geographic and climatic conditions' could be interpreted as the condition of biodiversity in the absence of pressures. Whilst the directive has a goal to phase out all pollution (Art 1.2), it is not considered feasible to remove all pressures on the marine environment. For instance it is probably not possible to eradicate invasive non-indigenous species and certain human activities by their nature give rise to some level of impact on the environment. To reflect these issues and to accommodate sustainable uses of the environment within the concept of GES, it was envisaged that some but unavoidable levels of deterioration would need to be incorporated into the definition of GES and its targets for Descriptor 1. Similar considerations can be applied for descriptors 4 and 6."*

This text clearly makes the point that even sustainable use of the marine natural resources must entail some trade-off in respect of biodiversity loss. The figure following this text in the manual, copied here as Figure 9.2.1, illustrates the types of deterioration that that could occur in respect of different MSFD descriptors. It also makes clear the boundary of allowable change; that point at which further deterioration takes the ecosystem below good environmental status: indicative of non-sustainable overexploitation. This figure implies that some reduction in the population abundance of sensitive species is acceptable: the situation illustrated in Figure 9.2.2 A and B. However, basic population dynamics dictates that sensitive species are less able to sustain a given level of mortality.

The OSPAR diagram (Figure 9.2.1) implies that the loss of sensitive species is deemed unacceptable; species loss represents a situation considered below GES. Populations are stable in size when mortality rates balance reproductive rates (ignoring the roles of immigration and emigration). If human activities increase mortality rates, the most sensitive species, those with the lowest reproductive capacity, suffer unsustainable mortality first. Population abundance of the most sensitive species will decline, trending towards extinction points, before serious declines in less sensitive species occur. Rising mortality rates, the consequence of increasing human pressure, will affect an increasing number of sensitive species, raising the number of species facing extinction risk (Le Quesne and Jennings, 2012). The actual situation is illustrated in Figure 9.2.2 C and D. In determining the trade-off between exploitation of natural marine resources and the consequent biodiversity loss, it is not a case of deciding how far we can permit the population abundance of all sensitive species to decline; rather it is a case of deciding how many of these sensitive species we need in order for ecosystem function to be maintained, and how many therefore we can afford to lose.

In a attempting to manage towards recovery, the same logic applies. It is unrealistic to assume that recovery of all depleted sensitive species can be achieved while still maintaining some socially acceptable and necessary level of exploitation. To recover all sensitive species would require the abolition of almost all human activity in the marine environment, and the socio-economic cost of this would be unacceptably high.

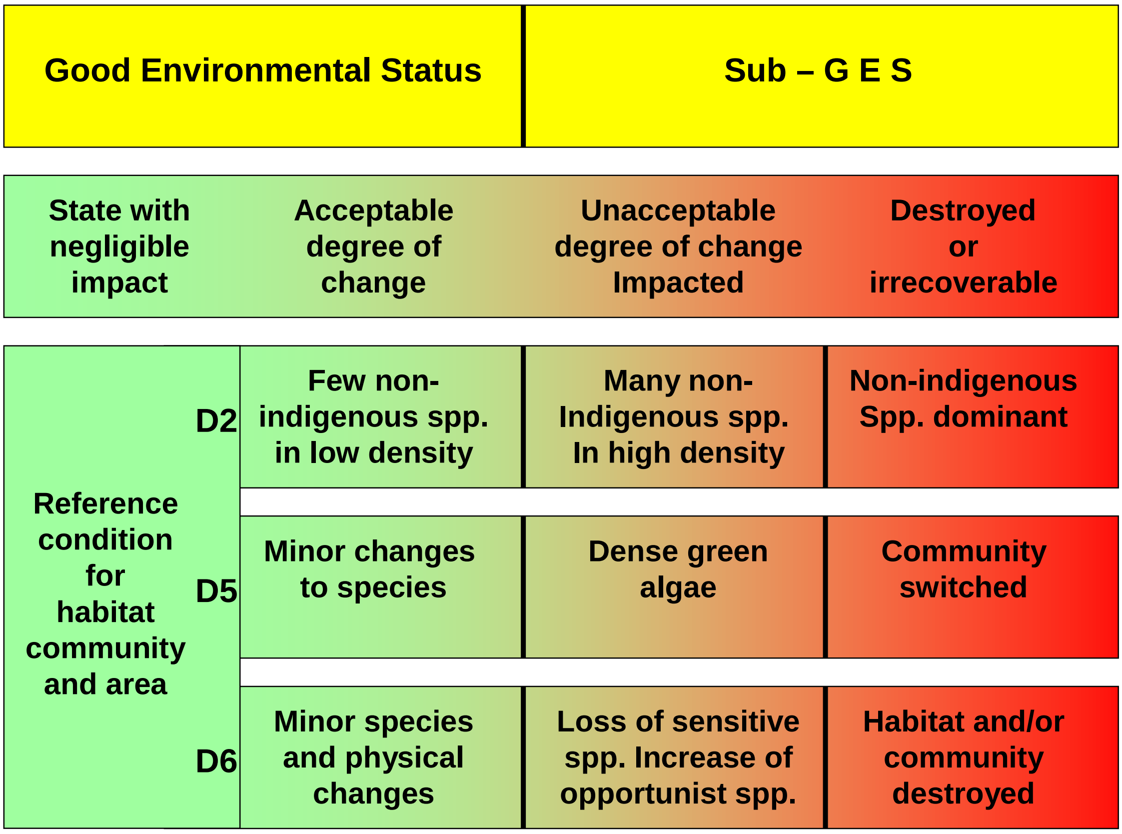

Figure 9.2.1. Reproduction of Figure 4 in in "OSPAR's MSFD Advice Manual on Biodiversity" illustrating the relationship between quality of a biodiversity component and changes caused by different pressures. Types of change are illustrative for the three Descriptors shown (D2: Non-indigenous species; D5: Nutrient enrichment (eutrophication); and D6: Physical disturbance to seafloor integrity).

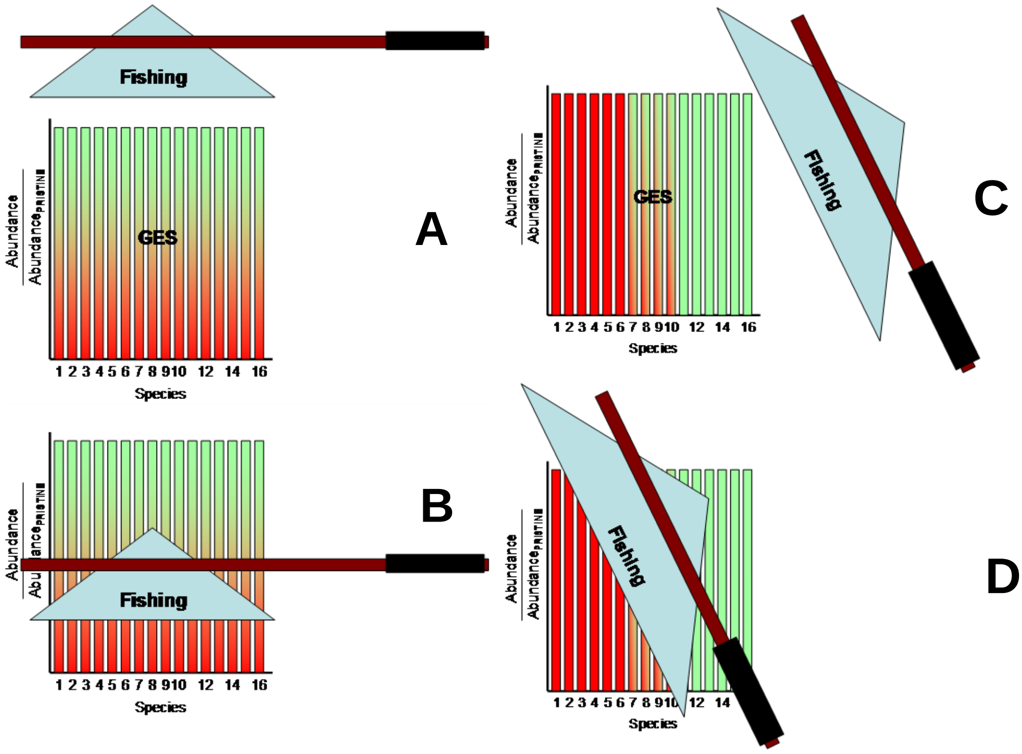

Figure 9.2.2. How the axe falls. A and B illustrate the situation assumed in the OSPAR Biodiversity Manual, where pressure reduces the abundance (expressed here as a fraction of the theoretical abundance in the undisturbed state) of "sensitive" species approximately equally. All species may eventually be reduced to a population size deemed to be below GES, but no species are lost. C and D illustrate the more realistic situation, where increasing pressure reduces the populations of the most sensitive species to near extinction point.

9.3 Economic and social indicators

We tabulate economic and social indicators applicable for inclusion into ecosystem modelling and management (Table 9.1.3.1). To fit these into a descriptor/indicator framework we adopted that used by an EC 7th Framework project, Making the European Fisheries Ecosystem Plan Operational (MEFEPO; Piet *et al.*, 2011). They considered two descriptors for the economic component (efficiency and stability) and three for the social component (community viability, job attractiveness and food security). Against this framework we introduced indicators from the literature that has been used to address fisheries issues, and those described in the Data Collection Framework (DCF) for the Common Fisheries Policy. Our selection of fit to Descriptor is not meant to be prescriptive but rather reflective of their use to date. We have tried not to be repetitive in this exercise; however we chose to present similar but different indicators due to the fact that data availability may limit options in some areas (cf. Piet *et al.*, 2011). Some of these indicators are more appropriate to large-scale applications, while others can be calculated over a range of spatial scales (cf. Franquesa, 2001). Matching these indicators to the appropriate sea area will require integrating information on métiers at a local-scale and building to higher levels of aggregation. From this table, indicators can be chosen to quantify the socio-economic objectives in Section 9. To our knowledge the economic and social indicators have not been subjected to the same evaluation as the biological indicators have; that is following the Rice and Rochet (2005) or similar framework. For many we suspect that limits and reference points have not been developed (especially for the social indicators); however, depending upon the application they may not be necessary.

Table 9.1.3.1. Economic and social indicators.

Pillar	Descriptor	Indicator	Reference
Economy	Efficiency/ Profitability	Revenue based on market price (euro/kg) of demersal species/size category and assumed market price of other species (the effect of a change in size on market price was not included); Cost was determined by effort where cost of one unit of effort is calculated per fishery assuming break-even situation at the start of the time-series; Profit = Revenue – Cost	MEFEPO
		Net earnings, i.e. gross earnings (landings × prices) less running or variable costs (depending on effort)	Rochet *et al*., 2012
		Gross cash flow, i.e. net earnings less crew wages and other costs, considered as "nonvariable" operational costs	Rochet *et al*., 2012
		Full equity profit (gross cash flow less economic depreciation), divided by the amount of capital invested	Rochet *et al*., 2012
		Profit Rate (PR), which indicates the percent ratio of yearly net profits plus the opportunity cost in relation with the investment.	Franquesa, 2001
		Total value of landings: Sum total of weight landings per species multiplied by value of species per unit weight. Only considers total value of landings, does not taken account of operating costs (profit).	MEFEPO
		Income: Gross value of landings, income from leasing out quota, direct subsidies, other income	DCF
		Gross Estimated Profit (GEP), which indicates the total profits obtained by the whole of the vessel owners, once the operating costs have been deducted. Such costs include: Salary Cost (SC), Opportunity Cost (OP), Costs related to Fishing (CDxTD) and Yearly Fixed Costs (YFC).	Franquesa, 2001
		Costs: crew wages, imputed value of unpaid labour, energy costs, repair and maintenance costs, other operational costs, capital costs (annual depreciation), capital value (value of physical capital, value of quota), investments in physical capital	DCF
		Financial position: debt/asset ratio	DCF
		Net Estimated Profit (NEP), which shows the total earnings obtained by the whole of the owners, once the depreciation cost has been deducted from the GEP. This cost is calculated following the criterion that the shelf life of a vessel is ten years.	Franquesa, 2001
		Gross Added Value (GAV), which expresses the Added Value that the segment in question contributes to the National Economy. This includes: salaries, profits, opportunity cost and depreciations.	Franquesa, 2001
		Vessel Physical Productivity (VFP), shows the average production of each vessel in terms of weight of landings.	Franquesa, 2001
		Capacity Physical Productivity (CFP), indicates average production in terms of weight of landings for each capacity unit (GT) of the vessels.	Franquesa, 2001

Pillar	Descriptor	Indicator	Reference
		Power Physical Productivity (PFP), shows the average production in terms of weight of landings for each power unit (HP) of the vessels.	Franquesa, 2001
		Per vessel Hour Physical Productivity (HFP), indicates the average production in terms of weight of landings for each full fishing hour. The total fishing time (T) results from multiplying the number of fishing hours by working days and then by the number of working days in one year (TD)	Franquesa, 2001
		Capacity Productivity (PGT), shows average production in terms of market value in the first sale for each capacity unit installed (GT) in the vessels.	Franquesa, 2001
		Man Physical Productivity (MFP), shows the average production in terms of weight of landings for each man employed.	Franquesa, 2001
		Man Productivity (MP) shows average production in terms of value in the first sale for each man used.	Franquesa, 2001
		Cost of the management of the fishery (where industry pays for costs)	PIRSA
	Stability	An index based on the fluctuation (how often is there a change from increase to decrease or vice versa) and deviation (how much is the change) in terms of the yield	MEFEPO
		Value weighted SSBPR compared to SSBPRF=0: Landings value weighted average SSBPR in scenario compared to SSBPR predicted under a no fishing scenario. Assumes more stable landings can be taken from species at a higher SSBPR, and value weighted to give emphasis to commercially important species.	MEFEPO
		Fish prices by commercial species; Landing prices (LP) represents the average market price of landings.	Franquesa, 2001; DCF
		Fish Contribution to the GNP (FCG), shows the importance of fishing production in the Gross National Product.	Franquesa, 2001
		Ratio Harvesting Value (RHV), shows the importance of fishing in comparison to aquaculture in terms of income.	Franquesa, 2001
		Ratio Harvesting Weight (RHW), shows the importance of fishing in comparison to aquaculture in terms of production weight.	Franquesa, 2001
		Opportunity Cost (OP) shows the yields that the owner could obtain should he invest his money in National Debt instead of investing in his business. This means that the owner is relinquishing that potential income. There is a profit in its economic sense when the yields of the invested capital surpass the opportunity cost	Franquesa, 2001
	Other	Fleet Capacity: number of vessels, GT, vessel engine power (measured in kW), vessel age	Rochet *et al.*, 2012; DCF
		Invested capital (IC) shows the current value of the whole of the vessels.	Franquesa, 2001

Pillar	Descriptor	Indicator	Reference
Social	Community viability	Number of Jobs	MEFEPO
		Employment generated in man days employment at sea (does not account for shore based processing and maintenance employment).	MEFEPO
		Employment: engaged crew	DCF
		Vessel Productivity (PV), shows average production in terms of market value in the first sale for each vessel.	Franquesa, 2001
		Power Productivity (PP), shows the average production in terms of market value in the first sale for each power unit (HP) of the vessels.	Franquesa, 2001
		Per Vessel Hour Productivity (PVH), shows the average production in terms of market value in the first sale for each fishing hour.	Franquesa, 2001
		Ratio Fish Employment (RFE), indicates the ratio of employment created directly by the fishing industry	Franquesa, 2001
	Job attractiveness	This indicator utilized a proxy calculated by applying a multiplier based on the operational functionality of the gear to the effort per métier.	MEFEPO
		Average wage (AW) indicates the average salary obtained by each man employed by job class.	Franquesa, 2001
		Accident rate	MEFEPO
	Food supply	Total Yield	MEFEPO
		Apparent Consumption shows the gross consumption of fishing products per inhabitant. It can be expressed either as weight of consumed fish per inhabitant (WAC) or as expense per inhabitant (VAC).	Franquesa, 2001
		Fish Coverage Rate (CR), shows the rate of apparent consumption covered by the production (regional or national scale).	Franquesa, 2001
		Fish Commercial Balance (CB), shows whether exports or imports of fishing products are higher	Franquesa, 2001
		Extraversion Rate (DR), shows to what extent the fishing sector of a country depends upon foreign trade, both for imports and exports	Franquesa, 2001

9.4 Methodological approaches to handle trade-offs between biodiversity and socio-economic indicators

Several families of tools are available to handle trade-offs in a decision process. The simplest representation is a decision table consisting of a number of management options (columns) compared against a suite of biodiversity and economic and social indicators (rows). The cells of the table contain the expected value of each indicator for each management option. Alternatively, the indicator values can be graphed, as in the MEFEPO example below (see Section 9.5.1).

Mathematical concepts and tools are being developed to help formalize trade-offs when a dynamic model of the system is available. These tools will help quantify and illustrate the losses on one objective incurred for the sake of gains on another objective. A second family grouped under the name of "multicriteria decision analysis" has been developed in the social sciences and in management realms. These approaches seek to take explicit account of multiple criteria in helping decision-makers explore decisions that matter. In decision analysis, a decision problem typically is broken down into a set of smaller problems that are easier to address, and a formal mechanism is applied for integrating the results of the partial problems so that a course of action is indicated for the larger problem. Suites of structured questions are then asked to decision-makers, or stakeholders, to help formalize the decision process and make it more transparent. The third family seeks to get rid of the complexity introduced by handling multiple criteria, by expressing all dimensions of the problem into one single currency – monetary value. These are the valuation methods and the related ecosystem services. Below we present examples of each of these families of tools. We stress that these methods do not automatically lead to decisions; rather they are tools that can inform decision-makers and help them make their decision process and criteria more explicit.

9.4.1 Modelling trade-offs in dynamic systems

Two types of approaches have been developed to address multiple constraints in dynamic systems. **Optimization** seeks a "best" solution under a set of constraints, while viability theory just outlines the boundaries of decisions and states that are consistent with multiple constraints. Optimization consists of finding the management option that maximizes one criterion (e.g. an economic objective) under constraint (e.g. a conservation threshold). When there are several constraints, direct optimization is not possible, and one seeks the Pareto frontier (the system is on a Pareto-optimum if improvement of any one objective would reduce the performance of the others. The Pareto-frontier can be viewed as the expression of the trade-off between the different objectives. For example, Cheung and Sumaila (2008) used a mass-balance (Ecopath) model of the northern South China Sea (NSCS) to investigate the trade-offs between four policy objectives: maximizing economic rent, employment, and ecosystem integrity (measured by average longevity and functional group diversity), while conserving vulnerable species. The Pareto-frontiers they find exhibit different shapes, from linear (social vs. conservation) through convex (economic vs. conservation) to sigmoid (economic vs. ecological integrity, Figure 9.4.2.1). Since NSCS fisheries in the 2000s were suboptimal in achieving conservation and economic objectives, there should be room to improve conservation status without compromising the overall long-term economic benefits from the fisheries.

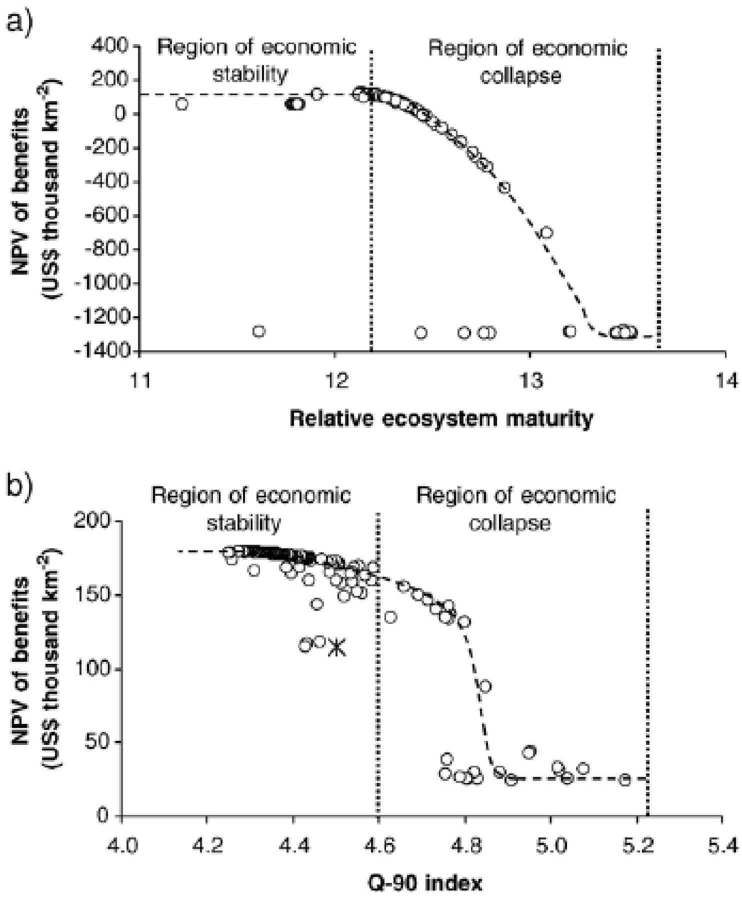

Figure 9.4.2.1. Trade-offs between the economic objective and ecosystem integrity, evaluated by running the policy optimization routine with different weighting factors for each objective over 30-year simulations. The outcomes shown are approximated Pareto-frontiers between the net present value of benefits of the fishery and (a) relative ecosystem maturity (the longevity-weighted summed biomass over functional groups in the model), and (b) Q-90 biodiversity index (indicates the diversity of the functional groups in an ecosystem, calculated as the slope of the cumulative functional group abundance curve between the 10 and 90 percentile. Higher Q-90 index indicates higher functional group diversity). High economic rent levels are incompatible with a high degree of ecosystem integrity; the trade-off is smoother for longevity than for biodiversity.

Viability theory is an alternative way of handling multiple constraints, related to the control of dynamic systems. It consists in defining sets of controls (management decisions) that will keep the system within a set of constraints (potentially conflicting management objectives). Sustainability requires both a set of system states meeting the set of constraints; the viability kernel, and a set of "viable" controls, to be non-empty. Note that there may be several viable controls, leaving room for different management decisions within the allowed set. In this approach, trade-offs might be expressed as the change induced by a change in one constraint on the size of the viability kernel; that is, in the potential realization of another constraint. Cury *et al.* (2005) suggested viability analyses as a means to operationalize an ecosystem approach to fisheries management. For example, the economic constraints (fuel costs and subsidies) on the French Guiana shrimp fishery recently changed and are affecting the local Frigatebird population, which used to feed on discards from this fishery and now starve owing to reduced activity. Martinet and Blanchard (2009) used the viability approach to examine the trade-off created by this change in economic constraints on the conservation objectives for the Frigatebirds (Figure 9.4.2.2). If the economic context becomes more difficult, the maximum viable effort decreases, inducing a lower potential conservation of the bird population. This approach also works with stochastic dynamics; in that case the viability kernel is defined by a high probability of viability. Using this approach Doyen *et al*. (2012) outlined system subsets in which either hake and *Nephrops* stocks are both conserved within sustainable (precautionary) limits, or the fleets that exploit them have all positive profits; the domain satisfying all sets of constraints is smaller than just the intersection between the two subsets, illustrating the trade-off.

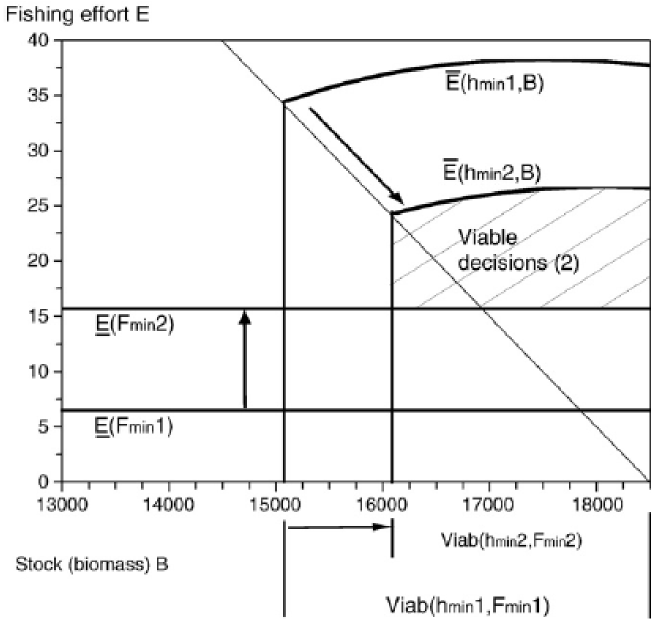

Figure 9.4.2.2. Sensitivity of the viability kernel (hatched) to the constraint levels determining catch per unit of effort (h_{min}) and conservation (F_{min}, minimum number of bird pairs able to feed chicks). The system is viable (i.e. will satisfy the constraints for ever) if the shrimp stock biomass is sufficient, and if fishing effort is neither too high (to allow sustained shrimp harvest) nor too low (to generate discards enough to feed the birds). If costs increase and/or subsidies decrease, the minimum catch per unit of effort necessary to keep the fishery profitable increase. This will reduce the viability kernel "top–down" from $\bar{E}(h_{min}1,B)$ to $\bar{E}(h_{min}2,B)$. On the other end, if conservation objectives are strengthened and the target number of bird pairs is increased, the minimum level of fishing effort to feed these birds increases, reducing the viability kernel "bottom–up" from $E(F_{min}1)$ to $E(F_{min}2)$. If the economic context is harsh and the conservation objectives ambitious, the viability kernel may be empty; the system could not satisfy both constraints in the long term. In that case the conservation objectives need to be softened, or subsidies to maintain the birds can be fed into the system (from Martinet and Blanchard, 2009).

9.4.2 Multicriteria decision analysis

We present two examples of this family here: goal programming and choice experiments. Many more tools have been developed and used in various settings, including fisheries management.

Goal programming is used in problems with very large or infinite numbers of decision alternatives, and can be viewed as algorithmic implementations of a natural human decision-making heuristic "satisfying". For each criterion in a decision, and each alternative, a deviation is defined as the difference between the goal for this alternative and the alternative's score on this criterion. The typical implementation of goal programming then minimizes the deviations over all criteria. Goal programming approaches were used to investigate trade-offs among employment, income and profits of cod fisheries in the Barents Sea (Leung, 2006). Mardle *et al*. (2001) explored trade-offs in the North Sea demersal fisheries between four goals, i.e. maximizing long-term profit, maintaining employment, minimizing discarding and minimizing deviations from the current division of quota allocations between fishing nations. An extension of this study (Mardle and Pascoe, 2002) included a fifth goal, maximizing short-term profit. Uncertainty in model parameterization was addressed with stochastic simulations, the result of which indicated that many of the solutions of the goal programming optimization were sensitive to the assumptions in the underlying operational model. Nevertheless, the authors conclude that a compromise solution can exist between short-term and long-term objectives in the management of the North Sea demersal fisheries. They highlighted the ease with which alternative scenarios and stakeholder preferences could be investigated, where straightforward weight modification on the goals provided a "What if" scenario analysis framework, which could aid discussion of proposed policies.

In a **Choice Experiment** (CE) framework, choices are broken down into component attributes, which are presented to respondents normally as set combination of the attributes. Respondents are then presented with a sequence of these choice sets, each containing alternative descriptions of, in this case, the fisheries management choices, differentiated by attributes and levels. Respondents are then asked to state their preferred alternative within the hypothetical choice set.

By observing and modelling how respondents change their preferred option in response to the changes in the levels of the attributes, it is possible to determine how stakeholders would trade-off between the different management options. By including price/cost or change in income as one of the attributes of each option, the monetary welfare impact of moving from some status quo fishing policy to an alternative fishing policy with attribute levels set to be representative of what would result under alternative management strategies can be calculated.

Table 9.4.3.1. A purely hypothetical example of a possible choice card.

	Option A	Option B	Option C
Technical Measure	TAC	Standard Quota	Tradeable Quota Permits
Employment	neutral	Increase	Decrease
Impact on Biodiversity	Negative	Neutral	Positive
Change in price per Tonne	+20	-10	0
Which do you like best			

Each stakeholder might be presented with eight to twelve of these choice cards, whereby the levels of each attribute change with each hypothetical management option. The stakeholder trades off one attribute vs. another when making his choice on each card. The data on choices made are then used in a decision-making, discrete-choice model in which the stakeholder's choice is estimated as a function of the attribute levels. The marginal value of a unit change in an attribute of the management option can be calculated as can the value of a package of attributes that make up alternative policy options. Data from the CE are analysed by employing the theoretical framework of random utility models (McFadden, 1974). It is assumed that the observed choice is the one associated with the highest obtained utility, U_{ni}, which is assumed to consist of a systematic part, V_{ni}, and a stochastic part, ε_{ni}:

$$U_{ni} = V_{ni} + \varepsilon_{ni} .$$

The probability that respondent *n* chooses alternative *i* from the set of *J* alternatives is given by:

$$P_{ni} = \text{Prob}(V_{ni} + \varepsilon_{ni} > V_{nj} + \varepsilon_{nj} \forall j \neq i) .$$

The observed utility V_{ni} is usually assumed to be linear in the parameters so that $V_{ni} = \beta x_{ni}$, where x_{ni} is a vector of observed variables relating to alternative *i*. If ε_{nj} is assumed to be an independently and identically distributed extreme value, this probability will have a closed form expression, leading to the conditional logit model:

$$P_{ni} = \frac{e^{\beta' x_{ni}}}{\sum_j e^{\beta' x_{nj}}} .$$

which is frequently employed by researchers to analyse choice data. The steps in conducting a Choice Experiment for application in a fishery management context might be as follows:

1) Decide on the management attributes to include in the choice sets (including how the price or cost coefficient should be specified).

2) Decide on the levels to use for each attribute.

3) Decide on number of cards to include in survey and come up with choice card design so as to optimize the combination of levels on the cards (orthogonal or Bayesian designs are popular).

4) Include choice cards in survey and interview stakeholders.

5) Analyse completed choice card data and run Random Utility Models.

9.4.3 Economic valuation

Monetary valuation of biodiversity allows for a direct comparison of the value of alternative management options, thus facilitating cost–benefit analysis of biodiversity practices. Economic valuation of biodiversity is based on an instrumental perspective on the value of biodiversity, a reductionist approach value, and starts from the premise that social values should be based on individual values (Nunes and van den Bergh, 2001). It is operationalized through explicit biodiversity changes, preferably marginal or small, and thus involving the design of alternative policy options or scenarios. In the absence of market price for biodiversity values, one can distinguish revealed preference and stated preference, including contingent valuation (CV), which is the most commonly used method. Most studies lack a clear perspective on biodiversity distinct from biological resources and fail to apply economic valuation to the full range of biodiversity benefits; therefore, valuation methods may provide lower bounds.

One component of economic valuation is benefits people obtain from ecosystems (e.g. food, timber, clean air, Millenium Assessment, 2005). The concept of **ecosystem services** (ES) conveys the utilitarian argument that nature and thus nature conservation contribute economic benefits to human welfare (Gomez-Baggethun *et al.*, 2010; Norgaard, 2010; Norris, 2012).

Two diverging methodologies to assess and develop strategies for the sustainable use of ES have evolved within ecological economics, i.e. the concepts of weak (WS) and strong sustainability (SS) with basically different understanding of the values of ES. This distinction has considerable implications for the link between economic and ecological sectors, and the understanding and measurement of sustainability. Thus, trade-off analysis with the aim to maintain sustainability is different between WS and SS with different sets of constraints regarding the maintenance of assets.

In WS, ecological resources are regarded as natural capital (Costanza *et al*., 1997; Daily 1997), and valuation of ecosystem services is an intrinsic component of global sustainable development programmes to support decision-making through cost–benefit analyses (TEEB, 2010; World Business Council for Sustainable Development, 2011). Ecosystem services can represent market (e.g. food) and non-market values (e.g. amenities) (Costanza *et al*., 1997). In neoclassical theory, value depends on the price obtained in an exchange process reflecting the interplay between marginal costs and marginal benefits. Goals of maintaining ecosystem services are fully internalized into the economic sector (i.e. 'mainstreaming' Cowling *et al*., 2008), implying that natural resources can be adequately managed by markets (van der Meer, 1994). As a consequence of internalization, ES have become commodities and thus tradable and substitutable units on a global market (Gomez-Baggethun and Ruiz-Perez, 2011; Peterson *et al*., 2009), untying them from problems of unconstrained economic growth (Norgaard, 2010). The valuation of market values is straightforward, and a set of established capital based tools to estimate non-market values exists (e.g. travel costs, hedonic pricing, contingent valuation, etc. (Beck and Helfand, 2011)). However, valuation methods for non-market ES are always biased towards value systems of those being surveyed to indicate the value (Barkmann *et al*., 2007). In SS related concepts, the economic sector is regarded as an open subsystem of the ecosphere (Gomez-Baggethun *et al*., 2010) and the link is seen as social-ecological coupling of equally important units (McLeod and Leslie, 2009).

The neoclassical definition of sustainability in WS is 'maintaining human welfare' regardless of the sources and ES that generate this welfare. Optimistic attitudes towards the possibilities to substitute benefits are at the heart of this WS concept (van der Meer, 1994) which ultimately allows diminishing natural capital as long as sufficient human-made capital is available to replenish any foregone human welfare (Howarth and Farber, 2002; Neumayer, 2010). An example of complete substitutability is the net positive impact concept in mining projects (Olsen *et al*., 2011), where a damage in one ecosystem good is balanced against improvement in another, both changes measured in monetary terms by means of economic valuation of ecosystem services. The anthropocentric view of WS has been contested in that properties reflecting the integrity of ecosystems are denied (Dobson, 2005; Peterson *et al*., 2009). This criticism is accounted for in the concept of SS not foreseeing interchangeability between different types of capital. A minimum of either type of capital, i.e. natural, economic and social, must be maintained to reach sustainability, i.e. the critical level of capital (Daly, 1990; Daly, 1992; Garmandia *et al*., 2010; Hediger, 1999). The FAO sustainability definition for capture fisheries relates to the 'strong sustainability' concept (FAO, 1999).

The SS approach implies independent assessments for each of type of capital, encompassing detailed mapping of natural features, the analyses of flows and impacts on habitats and habitat transformation in the assessment of natural capital (Cowling *et al*., 2008). Accordingly, SS implies more and stronger constraints in trade-off analysis than WS. In turn, a multitude of specific ecosystem indicators are required to measure the different types of natural assets instead of monetary values of ES as the sole currency to define trade-offs.

9.5 Examples of trade-offs between diversity conservation and socio-economic objectives

9.5.1 Modelling the relationship between biodiversity and socio-economic development

Clausen and York (2008) used cross-national data to examine how socio-economic development affects aquatic biodiversity. Data for fish species from >140 countries were used. A negative binomial regression model was fitted to the data to test the statistical hypothesis (the environmental Kuznets curve) that environmental quality exhibits a non-monotonic U-shaped relationship with affluence, commonly represented by GDP per capita. This relationship has been hypothesized to arise because of the emergence of an ecological rationality at high levels of affluence, resulting in environmental concerns spreading through society, such that political and socio-economic restructuring occurs that benefit the environment. In addition, environmental quality may be a luxury good; such that it is enhanced at high levels of affluence. In the model, aquatic biodiversity was measured by the number of threatened (according to IUCN Red List) marine and freshwater fish within a country's EEZ. A number of other biophysical variables were considered: total number of fish species, percentage of total fish species that are freshwater, area of freshwater habitat, area of continental shelf, percentage of total habitat that is freshwater, total land area and the number of non-endemic species. Socio-economic variables considered in the model were: average fish capture effort, average total population, average percent of population living in urban areas and average GDP per capita; the average of each variable was calculated for each country over yearly values in 1980–2000. The full version of the model included linear terms for all variables and a quadratic term for the average GDP per capita. However, the quadratic term for GDP per capita was found to be non-significant; only the linear term for average GDP per capita had a significant, positive effect. Thus, a U-shaped relationship between biodiversity and affluence was not supported. This study was thus able to provide evidence on whether the interaction between an ecological indicator and a socio-economic indicator is linear or non-linear.

9.5.2 Making the European Fisheries Ecosystem Plan Operational (MEFEPO)

Management trade-offs between ecological, economic and social indicators (see Table 9.1.3.1 above) were evaluated in an EC 7th Framework project, Making the European Fisheries Ecosystem Plan Operational (MEFEPO; Piet *et al.*, 2011). Stakeholders were involved in selecting and prioritizing the indicators used. MEFEPO specifically evaluated performance of biodiversity indicators (as per the MSFD) under five management scenarios (trade-offs) in two of their three case study areas (Northwest Waters, Southwest Waters). As each indicator was separately assessed, it is also possible to examine trade-offs among indicators with this approach. The management scenarios considered were:

- Business as usual (BAU): based on the current management situation, none of the three pillars in the management scenario is given particular emphasis;
- Achieve the minimum ecological requirements for Good Environmental Status (GES);

- Maximum long-term harvest (MSY): emphasis on ability to harvest food from the sea, strong link with the social pillar in terms of contribution to global food supply but to achieve a sustainable harvest of seafood there must be healthy commercial stocks;
- Maximum long-term employment (MEMP): emphasis on the social pillar and resilience of fishing communities;
- Maximum long-term profit (MPRO): focus on economic pillar and economic efficiency.

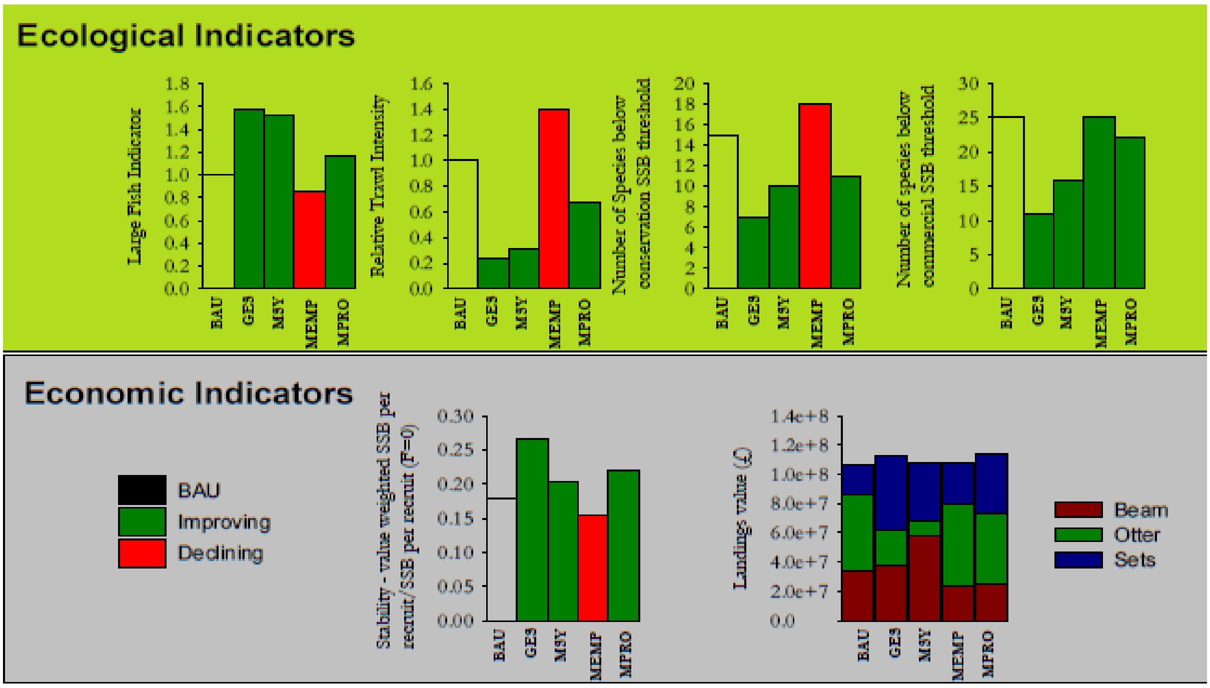

Figure 9.1.3.1. An example of how trade-offs among indicators and management scenarios were evaluated in the MEFEPO project (extracted from Figure 4.7; Piet *et al.*, 2011). This example shows two indicator suites from the Northwest Waters case study. In the original figure, performance was also evaluated against different effort scenarios of the fisheries examined. The indicator for the Biodiversity descriptor is illustrated in the second figure from the right in the upper panel (Social indicators not shown). Management options were: Business as usual (BAU); Achieve Good Environmental Status (GES); Maximum long-term harvest (MSY); Maximum long-term employment (MEMP); Maximum long-term profit (MPRO).

In the Northwest Waters a multimétier, multispecies model that simulated the multispecies 'technical interactions' of fishing operations was used that considered the demersal fish community and main demersal fisheries in the on-shelf Celtic Sea (Piet *et al.*, 2011). Biological interactions either between species or between species and their environment, and spatially explicit information were not considered. Data requirements for the model were: estimates of abundance at age, and landings and discard mortality by métier by species-at-age. Model outcomes were optimized for each indicator and management strategy in order to evaluate trade-offs among indicators and strategies (Figure 9.1.3.1). In the Southwest Waters data were not available for many of the indicators so expert judgment was used to make predictions on outcomes and results were presented in a similar format (Piet *et al.*, 2001). Interestingly, both approaches showed that the best management strategy for the biodiversity descriptor in both the Southwest and Northwest Waters was GES, followed by MSY and MPRO while the worst scenario was MEMP. This management strategy would maintain landings and high stability, implying that in this case the biodiversity and economic objectives are aligned. This approach converts the information used in decision trees into a graphical format, allowing for multiple indicators and outcomes to be viewed simultaneously.

9.5.3 Trade-off between benthic assemblages and fisheries MSY

Management solutions for trade-offs can be described formally as maximization of benefit Q based on decisions X_n affecting multiple pressures or management units, given a set of ecological or management constraints Y_m representing policy goals (Lackey, 1998):

$$Q = \max f(X_1,..,X_n \mid Y_1,..,Y_m)$$

This formulation strongly underlines that a quantifiable measure is required to deal with trade-offs and that this measure in chosen in relation to the constraints inflicting on the management.

In a case study undertaken by Fock *et al*. (2011), a production based risk criterion comparable to the MSY sustainability criterion in fisheries was employed to define a trade-off between (1) the state of benthic assemblages in order to meet the policy goals of favourable conservation status as defined in the Habitats Directive (HD-92/43/EEC) and good environmental status by MSFD, and (2) the goal delivered by the European fisheries policy to obtain sustainable fisheries under a maximum-sustainable-yield (MSY) regime.

The HD defines that "the conservation status will be taken as 'favourable' when … population dynamics data on the species concerned indicate that it is maintaining itself on a long-term basis as a viable component of its natural habitats, … ". MSFD defines, " 'good environmental status' means the environmental status of marine waters where … the use of the marine environment is at a level that is sustainable, thus safeguarding the potential for uses and activities by current and future generations. In line with this interpretation, the expert group on developing indicators for the MSFD descriptors (Cardoso *et al*., 2010) stated that there may be uses that do not cause serious adverse impacts, and the pressures from these uses cause such small perturbations that recovery is rapid.

Four different scenarios were investigated, i.e. the *status quo* scenario in terms of the 2006 distribution of fishing effort (Fock, 2008), scenario I as the closure of marine protected sites for large and small beam trawlers to satisfy HD requirements, scenario II as a fishing effort reduction scenario to meet requirements from the MSY target for plaice as one of the key target species in the area (Pedersen *et al*., 2009). Scenario III is a combination of scenarios I and II, i.e. effort reduction plus area closures. It was shown that only scenario III with two measures combined (i.e. effort reduction to MSY and areal closures) will likely meet requirements from three environmental policies, i.e. the Marine Strategy Framework Directive, the Habitats Directive, and the Common Fisheries Policy.

9.5.4 Depth-disturbance trade-off in continental slope fisheries

As part of the DEEPFISHMAN project (http://deepfishman.hafro.is) Lorance *et al*. studied several fish stocks that are exploited on the European continental slope at depths greater than 200 m, including hake, monkfish, and blue ling. Recent active debate has questioned both the sustainability of deep-water fisheries and the impact of bottom fishing gears on deep-water habitats. In addition to fisheries management aiming to manage the exploitation rate of the target species, management measures designed to protect the deep-water ecosystem have been taken or are under discussion. These measures range from fishing practices to reduce the impact of fishing on vulnerable habitats, to localized MPAs, to a complete ban on bottom fishing gears beyond a certain depth threshold. These management measures imply trade-offs that have not been explicitly described. For instance, it has been argued that fishing for demersal species should be banned in waters deeper than 200 m to reduce the impacts of fishing gears on deep-water benthic habitat, including cold-water corals, such as those reported between 200 and 800 m in the Bay of Biscay and Celtic Sea. However, on-board fishing vessel observation of the gillnet fishery for hake and the trawl fishery for monkfish show a bigger-deeper trend in the size of these species. Thus there is a trade-off between fishing deeper for larger fish and protecting deep-water habitats.

9.6 Summary and WGECO advice on handling trade-offs in biodiversity conservation

Socio-economic objectives are contained in the Convention of Biological Diversity and in conventions and directives related to biodiversity in the ICES area. Whether these objectives are implicit or explicit, they are coming to the forefront in research projects and management issues. In fact MSFD-related work now calls for economic and social analyses including ecosystem services and cost–benefit analysis. WGECO recognizes the importance of this work but also that it is a demanding task that will require clear policy guidance on objectives and science leadership on methodologies and information requirements.

Preservation of coastal fishing communities, and fisheries related employment in them, emerges as an important social objective and political priority. Making social objectives explicit can help to rationalize decisions that might otherwise seem economically inconsistent. For example, fewer, larger catch shares are typically more economically efficient but provide fewer employment opportunities. Differentiated management has been suggested for small-scale inshore fishing fleets.

There is no consensus on biodiversity targets, notwithstanding extensive work within the MSFD. Some biodiversity targets imply a pristine state; others recognize some alteration of biodiversity but not loss of species. But intensive fisheries and other human uses of the marine ecosystems have resulted in depletion of some species, which would be very difficult to recover without severely curtailing human uses. Even with biodiversity indicators and targets, decisions are ultimately be made by government bodies, so good governance is needed.

There are plenty of potential economic and social indicators for inclusion in ecosystem modelling and decisions involving biodiversity. Some economic indicators have obvious reference levels: e.g. the economic benefits of a particular marine management policy can be compared against the average benefits from the rest of the economy. Guidance is needed on which socio-economic indicators to use when evaluating management actions. How do socio-economic indicators map to the high-level objectives? Has the performance of these indicators been evaluated in a manner similar to ecological indicators? Is the covariation between different socio-economic indicators simple or more complex? WGECO is not aware that this work has been conducted yet, feels that a suitably qualified body should do it, but does not feel qualified to do the work on its own.

While it is convenient to view trade-offs in two dimensions, between an ecological outcome and an economic one, there are often several desired ecological outcomes, and multiple industry sectors with possibly their own individual economic goals. So the trade-offs are often multidimensional. Tools exist for evaluating trade-offs ranging from simple decision tables, trade-off models, multicriteria decision analysis, to economic valuation. Economic valuation works best for the economic pillar, but less well for the other two. Ecosystem services may not be the best approach for decisions involving biodiversity. Some of these approaches are developed as case studies but few have been applied to real fisheries-management decisions. WGECO feels that these tools should fully developed without being prescriptive about which methods are most appropriate.

Social equity and viable life-styles are difficult to put onto quantitative axes to contrast with profits, landed value, and typical economic indicators, and with population biomass or abundance, measures of ecosystem function, etc. These social considerations of equity, livelihoods, etc. are not captured well in most "willingness to pay" metrics. "Willingness to pay" metrics measure the preferences of the affluent sectors of society, who have the luxury of discretionary spending. Many participants in fisheries in which social objectives dominate economic ones may be economically disadvantaged to begin with, and have few employment alternatives. This makes formal trade-off analyses difficult to conduct, when the social dimension of sustainability is given equal status with the ecological and economic dimensions.

It would be convenient to define this issue out of the equation by arguing that "equitable sharing of benefits" is something of concern only to developing economies. Unfortunately that is not the case. Many ICES member states have maintenance of coastal communities and the lifestyles of those communities as at least an implicit, and sometimes an explicit, policy goal (Canadian East Coast, Norway and its coastal fisheries, other parts of Europe where maintenance of employment is a priority. In reality, trade-off analyses that fail to come to grips with "equity", "livelihoods", and "social well-being" will fail to be very useful in informing decision-making where the decisions are hardest (and often most urgently needed from both ecological and social perspectives).

Even if all indicators could be expressed in common currency, is one pillar more important to the other? When decisions are made, does socio-economics trump biodiversity? For example, there may be exceptions to achieving GES for reasons of overriding public interest or where costs are disproportionate.

In the MEFEPO example, it appeared that biodiversity and economic indicators were closely aligned, implying a mild trade-off. In the German Bight example, the multiple objectives could only be met with a combination of management measures, which implies a strong trade-off and highly constrained management options. There was no consistency in how trade-offs are handled. Guiding principles are useful but there is no best practice; the appropriate approaches and methods appear to be context specific.

Next steps? Broaden scope to include sectors and issues, in particular, nearshore areas (e.g. fjords) and land-based uses, where trade-offs may be particularly acute. WGECO would be happy to participate in a multidisciplinary framework involving economists and other social scientists.

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Annex 2: Technical minutes from the Vulnerable Marine Ecosystems Review Group (RGVME)

- RGVME
- Deadline for review 14 May 2012
- Participants: Margaret M. McBride, Norway (Chair), Leonie Dransfeld, Ireland, Angel Pérez-Ruzafa, Spain and Claus Hagebro (ICES Secretariat)
- Working Group: WGECO

Request 2: The bottom fisheries regulations implemented in the NEAFCRA are to be reviewed in 2012. In order to facilitate the revision ICES is requested to advice NEAFC on following issues

Note: WGDEEP has already reviewed and commented on WGDEC responses to Request 2. However, additional comments from RGVME are welcome.

a) Impact assessments

ICES is asked to propose elements to be included in impact assessments, required to satisfy the NEAFC bottom-fishing regulations in the NEAFC RA.

WGDEEP/WGECO response

ICES (2011a) discussed the potential applications of ecological risk assessment (ERA) methods. WGDEEP builds upon that previous work and examines a non-exhaustive list of (1) approaches to ecological risk assessment and (2) detect elements which are relevant to impact assessment in the NEAFC RA. Approaches examined included:

- Ecological Risk Assessment of the Effects of Fishing
    - Level 1: Scale-Intensity Consequence analysis (SICA);
    - Level 2: Productivity Susceptibility Analysis (PSA); and
    - Level 3: Sustainability Assessment for Fishing Effects (SAFE);
- The Marine Life Information Network (MarLIN) approach;
- The US National Research Council approach;
- Extended Overlap Models; and
- Population Level Models.

For each approach the analyses/assessments involved, data requirements, and strengths/weaknesses are examined and discussed.

In conclusion, reservations were expressed regarding types of approaches which create the potential for poor information (selection and weighing of the input parameters) to be presented as having more reliability than it actually does, and thus running the risk of producing false outcomes. For all approaches it is critical that decision steps be fully documented to maintain transparency and increase confidence in the outcomes.

RGVME comment(s)

- WGECO is to be congratulated for providing a very thorough analysis: 1) of the principles of ecological risk assessments, providing different examples which are being used, assessing their strength and weaknesses; and 2) applying some of the principles to the NEAFC request for impact assessments for new deep-water fisheries.

- The specific advice regarding the impact assessment for NEAFC needs to be added to the Advice, i.e. WGDEC Report (Pages 15–23).
- In the WGECO response to the request for impact assessment there seems to be some confusion as to what information is required from the applicant for the impact assessment, for example:
  - Information needed **before** any permission to fish is granted; and
  - Once the permission has been granted to conduct experimental fisheries, information to be collected by on-board observers during fishing operations (e.g. position coordinates, VMS data).
- To address elements 1 and 2 of the risk assessment relative to the fishery resource itself (fishing effort level, harvesting plan, and best available scientific information on current state of fishery resource) the report of WKLIFE (ICES 2012) could give very good supporting material on methods to develop reference levels for data poor stocks. There is a dedicated section on deep-water stocks in this report. In order to address element 2 on relevant habitat and biological information in the fishing area and element 3 of the risk assessment on the description of VMEs, the VME database from WGDEC would be a very important data source as well as other existing coral and VME databases (OSPAR?).
- Answers to the subsequent elements (4 to 6) outline in a very useful way steps that are needed, and methods that can be used to carry out the impact assessments. Where relevant, it draws attention to international guidelines for fishing in deep water, i.e. FAO.
- A flow chart working through the six elements of the impact assessment would be quite useful to be included in the advice. This would guide the reader through the different approaches that are proposed under each element.

b) Encounter thresholds

ICES is asked to assess the appropriateness of the current quantitative thresholds of VME indicator organisms, i.e. live coral and sponge, adopted in the NEAFC bottom-fishing regulations. The assessment should include an evaluation of the likelihood of achieving conservation objectives, i.e. the prevention of significant adverse impacts on VMEs as defined in the FAO guidelines.

WGDEC response

Current quantitative encounter thresholds for VME indicator taxa in the NEAFC area are 60 kg of live coral or 800 kg of sponge landed on deck per tow. Current experience with these thresholds is that no reports of encounters have been received (NEAFC, pers. comm. 2012). WGDEC believes that current thresholds are too high and that a 50% reduction to 400 kg (sponges) and 30 kg (corals) would better reflect the likelihood that a VME was encountered.

RGVME comment(s)

- We agree with the WGDEC proposal for a 50% reduction in threshold levels, but it is not enough. As available information is very scarce, in addition the presence of indicator species below the established thresholds and their abundance should be reported independently.

- It is important to map the distribution, scales of variability, and patterns in abundance of VME-indicator species. Frequency of occurrence could also be used as an indicator for relevant indicator species that occur only rarely or in low biomass/abundance.
- It is also important to incorporate fishing effort, and zero occurrences VME indicators in the data. Information on the hauls (location, number, duration, depth, etc.) should be reported for all hauls indicating, expressly, not only the presence and abundance of indicator species, but also their absence.

c) Move-on-rule

- ICES is asked to assess the appropriateness of the current move-on-rule adopted in the NEAFC bottom-fishing regulations. The assessment should take into account the different habitats where bottom fisheries occur, e.g. continental slopes, mid-ocean ridges and seamounts, as well as the variable amount and quality of information on the relevant spatial distribution of VMEs. The provisions inherently assume that a significant proportion of known and unknown VMEs have been protected by bottom-fishing closures and other regulatory measures aimed to achieve sustainable bottom fisheries.

WGDEC response

Encounter provisions, including the move-on rule, currently in force apply in both 'existing fishing areas' and 'new fishing areas'. Currently no encounters have been reported; hence the move-on rule has not been triggered and no temporary closures implemented. No applications for exploratory fisheries (in 'new fishing areas') have been received. Thus, there is no experience with the currently adopted move-on rule, and there is limited basis for discussing its appropriateness other than theoretical considerations.

Fishing vessels are required to record the location and bycatch of VME indicators even at levels below move-on-rule thresholds, thus logbooks may be a valuable source of data on the quantities and spatial distribution of VME indicators in normal fishing operations with bottom gears in the NEAFC regulatory areas. To date WGDEC has not requested access to these logbook data, and has not analysed incidence and quantity of VME indicators in relevant fisheries. WGDEC therefore recommends that NEAFC consider making vessel logbook data available to ICES. Until quantitative analyses can be conducted to provide a scientific basis for move-on distances, only theoretical factors can be considered to assess their appropriateness.

Discussion is presented on appropriateness of the move-on rule: in different habitats; for different fishing gear types; and in existing and new fishing areas. General consideration of an alternative move-on rule for longline vessels, and how the current move-on rule might be modified or an alternative one developed, concluded that:

- The ideal move-on rule would take into account the spatial size and shape of the VME encountered, VME patchiness, and the level of precision of encounter location positioning;

- The theoretical move-on distance should at least be equal to the longest dimension of the VME feature plus the spatial uncertainty of the reported encounter; and
- The biggest hindrance to progress is the lack of information on the size, shape and patchiness of most VMEs within the NEAFC regulatory area.

RGVME comment(s)

- RGVME agrees with WGDEC's response.
- It is cause for concern that since implementation of the move-on rule (in 2006?) no fishing vessels have reported VME encounters that have triggered temporary closures.
    - It calls into question the effectiveness of the on-board vessel observer programme which would be expected to eliminate or minimize questions about whether or not encounters with VME indicators (of any size) occur and are reported. If observers are not deployed on all vessels in existing fisheries, perhaps a stratified random design based on vessel type and area fished could be implemented to ensure a representative coverage of fishing activities, and minimize costs.
- In the absence of other data sources to help identify and determine the location of VMEs, it seems entirely appropriate to implement a programme requiring all vessels to submit their logbooks. Additionally if observers are on board, their duties might include monitoring and verifying the accuracy of VME indicator encounter reporting in that vessel's logbook.
- RGVME agrees with WGDEC's opinion that the encounter thresholds are too high, and should be reduced by 50% to evaluate this policy's effectiveness. Also,
- Specific conditions for the Move-on rule are complicated. To encourage cooperation and better reporting of VME encounters, it might be useful to implement a programme designed to educate/inform fishermen in the NEAFC regulatory area about the fundamental principles of benthic ecosystem ecology upon which the Move-on rule is founded, the function that VMEs play in fisheries productivity and sustainability, why closures are imposed to protect VMEs as well as the rationale, goals and objective, and requirements of the rule.

d) Alternatives to thresholds and move-on rules

ICES is furthermore asked to inform NEAFC on alternative or additional measures to the currently adopted encounter thresholds and move-on-rule, especially technical measures, that may reduce the risk of encounters with VME indicators.

WGDEC response

Every effort should be made to avoid fishing with bottom-touching trawls or bottom-set longlines in areas where there is a high likelihood of encountering VME indicators.

Current NEAFC measures, while appropriate to some existing fishing areas that exploit generally sedimentary seabed habitats, are considered inappropriate to fisheries operating in rugged terrain (e.g. seamount summits and edges, canyons, and rocky outcrops) where the densities and distribution patterns of VME indicators are poorly known.

Additional measures should facilitate enhanced precision in recording encounters and/or reduced likelihood of encounters. This is especially important in exploratory fisheries in 'new fishing areas' but remains relevant in 'existing fishing areas'. Lessening the risk of significant adverse impact can be achieved either by the use of fishing gear that reduces the extent of bottom contact, or by restricting the use of bottom-contact gears in certain areas where the likelihood of encounters is high. In the NEAFC regulatory area, fisheries are conducted mainly by trawls and longlines, and it is important to try to reduce bottom contact in both these gear types.

Minimizing gear impacts

Closely monitored bathypelagic trawling (just above the seabed) should be encouraged as an alternative to traditional bottom trawling whenever feasible. There may also be potential to design trawl doors/otter boards that have a reduced bottom contact or have a lower impact. Reduction in sweep length (wire between trawl doors and net) may also be a means of lessening the area impacted by the gear. WGDEC suggest that the question would benefit from consideration by the ICES Working Group on Fishing Technology and Fish Behaviour (WGFTFB).

While the impacts of longline fishing on VMEs are considered minor compared with those from traditional bottom trawling, longlines also have bycatch of VME indicators that can be cumulatively significant. Effort should be made to reduce bycatches. A technological alternative is to deploy vertically suspended longlines that are attached to the seabed with a single anchor. Vertical longlines are expected to have significantly less bycatch of VME indicators than bottom-set longlines.

High-tech monitoring and mapping

Virtually all vessels use advanced echosounders (and sometimes multibeam), gear geometry monitoring equipment, and advanced chart plotting software. In addition to the established requirement for VMS and logbook recording, it might be implemented as an additional requirement that vessels keep:

- Electronic records of bottom contact and exact tracks of individual tows -- at least in exploratory and seamount fisheries. This would facilitate documentation of occurrence of VME indicators at a relevant spatial scale, and document whether or not adverse impacts are occurring.
- Videos from individual tows and plotter tracks submitted together with catch and bycatch information would facilitate subsequent analyses of VME indicator records and assessments of whether or not a VME was encountered.

Taken together such information would provide strong evidence as to whether the fishery is having any significantly adverse impacts on VMEs.

In exploratory fisheries (in new fishing areas where observers are required) it is recommended to introduce pre-fishing mapping (ideally multibeam) as a requirement before the gear is deployed.

NEAFC may consider introducing some or all of these requirements in its 'Exploratory Bottom Fishing Protocol for New Bottom Fishing areas' and the 'VME Data Collection Protocol'. NEAFC might furthermore consider the utility of requiring a more extensive impact assessment requirement such as introduced in NAFO.

For those areas that have already been extensively mapped (mostly national efforts), it is strongly advised that data mapping fishing areas and associated VMSs be made available to vessels from all contracting parties.

Alternative management options for seamount fisheries

All seamount fisheries should be prepared to prove that their operations are not causing significant adverse impacts on VMEs through high-technology habitat mapping and fishery monitoring. Evidence should be provided that;

1) Current fishing practices are focused within existing trawled areas (based on logbooks, fishing tracks, etc.);

2) These areas are mapped precisely (sonar/multibeam data) at a fine spatial resolution);

3) These areas do not contain VMEs (net-mounted camera evidence

4) Vessels have the technology and experience to keep their fishery precisely within the existing fishery footprint (gear monitoring sensors, skipper's experience).

If a VME encounter occurs, regardless, there may be no alternative but for that vessel to move off that seamount and a temporary closure be enforced.

RGVME comment(s)

- RGVME agrees with these proposals. To further minimize impacts, more emphasis should be placed on establishing of a network of no-take zones where fishing is completely prohibited. Additional mechanisms to improve enforcement and ensure that fishing prohibitions are adhered to should be proposed.
- WGDEC provides an excellent account of alternative methods/approaches which can be used relative to thresholds and move on rules. Their emphasis on mitigation, avoidance, and thorough pre-planning of fishing trips to minimize impact on VMEs provides a good basis for advice issued from this request.

e) Identifying vulnerable marine ecosystems

ICES is furthermore asked, using the best available scientific information including bio-geographic information, to identify in the NEAFC Regulatory Area:

- Areas where VMEs do not occur;
- Areas where VMEs are not likely to occur;
- Areas where VMEs are likely to occur;
- Areas where VMEs are known to occur.

WGDEC response

The majority of VMEs are considered to be patchily distributed (aggregated in space) as they tend to be found in association with particular physical, environmental and hydrographical conditions and geological or topographic features. This is likely to apply to VMEs such as cold-water coral reefs, deep-sea sponge aggregations, and hydrothermal vents -- although conditions will vary regionally.

Variable certainty of different information sources

Information on the presence and absence of VMEs derives from a number of sources including visual surveys, trawl bycatch, fishing effort analyses, geo-physical surveys and predictive habitat models. Each of these data sources has its strengths and weaknesses in addressing these questions. It is important to appreciate both the degree of certainty that can be ascribed to each information source and the appropriate geographical scale that the information is most meaningful to managers (Figure 18):

- Visual surveys (drop camera, ROV observation, submersible, towed camera) are the most reliable source of information for the absence of VMEs, although the areal coverage in such surveys tends to be small. If large areas can be covered, for example, by using towed video cameras, a survey design may be possible in which interpolation across broader areas becomes possible.
- Trawl bycatch data are a less reliable source of absence data due to the fact that trawls are designed to catch fish not fragile, benthic organisms. Nevertheless a large number of tows in the same area that have zero bycatch can be taken as evidence of absence.
- Fishing effort analyses can be used an indirect method to infer the likelihood of absence of VMEs. If an area has been systematically trawled for many years it may be fairly safe to assume that either no VMEs were ever present, or that if they were present they have by now been destroyed. However, this cannot be taken as certain, unless the data are resolved to a geographical scale fine enough to assess precisely whether enough ground was left unimpacted to allow some remnants of VME to persist.

Geophysical surveys, such as sidescan sonar and backscatter analysis can potentially identify areas of VMEs at very fine scales as well as indicate where VMEs do not occur. Such information again tends to be on small geographic scales and while of use in delimiting local closure boundaries cannot give broad scale definition of areas where VMEs do not occur. Such information is only able to detect the presence of VMEs that sit above the seabed (such as *Lophelia* colonies or large sponges) but is unlikely to detect VMEs such as soft-bottom coral gardens.

Multibeam echosounder data from acoustic surveys are a rich source of information for seabed morphology, habitat mapping and classification. Seabed morphology has been shown to play a crucial role in the distribution of benthic biota in recent years.

Predictive habitat models or habitat suitability models (HSMs) have become an important new source of information in the debate over VMEs. At the global scale they are useful to identifying broad ocean-basin-scale areas where different types of VMEs are likely to occur. However, they currently are unable to discern fine scale features like small mounds, iceberg plough marks, small scours which are associated with presence of some VMEs. This is a critical shortfall because such fine scale features may combine at the regional scale to represent areas of greatest habitat significance.

Consideration was given to how the information sources above can in combination inform the four questions posed:

Areas where VMEs do not occur

Visual survey records showing non-occurrence of VMEs are available for some areas. Geophysical surveys -- using technology such as multibeam or sidescan sonar on automated submersibles -- can identify local areas that do not contain certain types of

VMEs such as *Lophelia pertusa* reefs, however they cannot be expected to identify all types of VMEs, e.g. coral gardens; it is not of sufficient spatial resolution to provide conclusive evidence of a lack of VMEs. Trawl bycatch data as noted above cannot be considered to give an absolute assurance of absence of VME. Predictive habitat modelling can potentially identify areas where VMEs do not occur, but cannot give 100% assurance of absence of VME. However the new generation of terrain-based habitat models may be able to resolve habitat suitability at a geographical scale that will be useful to managing fisheries better with respect to VMEs. In sum there is unequivocal evidence of areas of seabed where no VMEs are present, but these areas tend be rather small in area due to limitations in surveying the deep sea.

Areas where VMEs are unlikely to occur

Output from habitat suitability modelling (HSM) is useful because it gives a high degree of certainty where corals do not occur. Trawl bycatch data can be used in conjunction with other sources to indicate potential absence of VMEs. Analysis of fishing effort patterns can also be informative; areas consistently trawled for many years are likely not to contain VME species. In combination these data sources can be used to infer where VMEs are unlikely to be found.

Areas where VMEs are likely to occur

Most records used to assess the presence of VMEs are in fact records of species that may indicate the presence of a VME, not an actual observation of a VME. Thus, most of the data used in scientific advice is actually most appropriately used to infer where VMEs are likely to occur. This applies to trawl-survey bycatch data, and often to visual surveys of occasional observations of VME indicator species. It is also the most appropriate inference to be taken from published predictive habitat models.

Areas where VMEs are known to occur

There are relatively few records of known occurrences of VMEs, relative to the amount of VME indicator species data. Only visual or geophysical surveys (and ideally a combination of the two) can unequivocally demonstrate the presence of a VME.

Conclusions

Many data indicate the presence of VMEs throughout much of the existing NEAFC regulatory area. Fewer data provide unequivocal evidence of areas that contain VMEs. Where there is strong evidence of the presence of VMEs, areas have been closed to bottom contact fishing. But, not all potential VME areas are protected. The amount of information needed to unequivocally demonstrate the presence of VMEs often requires multidisciplinary research with information from: multibeam, side-scan, geo-physical analysis; boxcore sampling; visual surveys/ROV; habitat maps; and predictive models with confidence limits.

To be a useful tool to delineate VME boundaries, HSMs should be generated using terrain parameters derived from high resolution multibeam generated bathymetry. Given that multibeam data may often be collected by the fishing industry prior to fishing in areas of complex terrain, collection and sharing some additional bottom-trawl video/photographs from such areas could provide the basis for collaboration with habitat suitability modellers. The resulting topographic maps and predicted occurrence of VMEs could facilitate spatial zoning of safe areas to deploy fishing gears, and VME areas to be avoided.

RGVME comment(s)

Taking into account that VME indicators are primarily filter-feeders, trends in levels of abundance in these communities must be viewed in association with bottom currents. Therefore, modelling these currents with consideration of bottom topography and other habitat descriptors like sediment type, habitat complexity, primary productivity in the water column, etc. could provide a powerful predictive tool.