

ICES WGECO REPORT 2015

ICES ACOM COMMITTEE

ICES CM 2015\ACOM:24

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

8–15 April 2015

ICES Headquarters, Copenhagen, Denmark



International Council for
the Exploration of the Sea

Conseil International pour
l'Exploration de la Mer

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Recommended format for purposes of citation:

ICES. 2015. Report of the Working Group on the Ecosystem Effects of Fishing Activities (WGECO), 8–15 April 2015, ICES Headquarters, Copenhagen, Denmark. ICES CM 2015\ACOM:24. 122 pp.

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Executive Summary

The 2015 meeting of WGECO was held at the ICES HQ in Copenhagen, Denmark from 8–15 April 2015. The meeting was attended by 16 delegates from 11 countries, and was chaired by Anna Rindorf (Denmark). The work conducted was centred on 4 Terms of Reference concerning indicators of Good Environmental Status (GES) of the benthic community, the potential effect of a landing obligation on the benthic ecosystem, indicators of Good Environmental Status of Distribution and data weaknesses encountered as well as two additional topics judged by the working group to be of immediate importance to ecosystem advice in the light of the developments in the previous year: surveillance indicators and sensitive species.

WGECO continued the work to **develop and assess indicators of Good Environmental Status of the benthic community**. The task was centred on identifying indicators of GES for seafloor integrity (Descriptor 6). Indicators for fishing pressure were calculated for the Greater North Sea. Based on this, two complementary approaches for the calculation of impact indicators were presented. The two approaches illustrated how fishing affects the functioning of the seafloor involving bioturbators and surface depositors and how a mechanistic understanding of the fishing impact on the benthic community can be used to set boundaries and assess the status of the seafloor in relation to GES. A method to derive the relationship between various pressures, i.e. other than fishing, and ecosystem functioning was proposed. The method disentangles (1) the effects of the pressure on the benthic community determined by the sensitivity of the benthos to that pressure and (2) how the affected benthic community, with a different species-composition will display a change in its functioning.

Work on **evaluating the effect of a landing obligation on the benthic ecosystem** was continued through the identification of non-commercial key scavengers on discards in the benthic communities. Studies investigating the aggregation of organisms around discards, presented to them as bait were reviewed and scavenging organisms identified as species which were able to detect discards and move towards them for consumption, and are hence defined over a continuum between those that are close to obligate scavengers through to predators that will occasionally scavenge. Baited traps and cameras were the main sampling methods to study the consumption of discards on the seabed, and determine the abundance and diversity of species retained. WGECO proposes to investigate the effect of the landing obligation on these species by developing three possible indicators: (1) the proportion of benthic biomass which are key scavengers, (2) the biomass or abundance of key scavengers and (3) indicators based on important areas of spatial overlap of key scavenger species distribution and discards from main fleets.

Indicators of Good Environmental Status of Distribution in the context of the Marine Strategy Framework Directive (MSFD) were discussed through an identification of the challenges in application of distribution indicators and an examination of the attributes of distributions which should ideally be monitored, and their value to management. Challenges identified included the sampling effects particularly with relation to rare species, the complexity of metrics currently available to assess distributional changes and the geographical scales needed to assess distributions which do not fit the MSFD regions for many species. The approach was demonstrated in an analysis of a relatively rare species, cuckoo ray, in the continental shelf waters of the Celtic Seas and Bay of Biscay MSFD regions. Visual exploration of the distribution of Cuckoo ray allowed attributes of the distribution that are of scientific and manage-

ment interest to be determined and indicators that accurately reflect changes in these attributes were suggested.

Surveillance indicators may have an important role in implementation of the MSFD. They are used to monitor key aspects of the ecosystem for which there is (i) insufficient evidence to define targets and support formal state assessment, and/or (ii) where links to anthropogenic pressures are not sufficiently understood to underpin specific management advice. Surveillance indicators are not expected to directly track state in relation to GES, but may provide complementary information (including warning signals) that informs and supports science, policy and management. WGECO presents (1) a framework for including surveillance indicators into the Activity-Pressure-State-Response (APSR) process, (2) examples of how surveillance indicators can inform GES assessment and fish stock assessments, and (3) criteria for selecting useful indicators. Surveillance indicators may have an important role in the MSFD and merit further investigation.

Sensitive species are a key component of ecosystems and form an important component of biodiversity (MSFD Descriptor 1). An indicator of sensitive fish has been developed by the Working Group on Biodiversity Science (WGBIODIV) and taken up by OSPAR, and this led the group to a general review of the appropriate methods to identify species sensitive to pressures, the estimation of exposure and the combination of these aspects in a risk based advice framework. Worked examples describing the identification of fish and benthos sensitive to fishing pressure are provided and a potential format for advice on sensitive fish species is suggested.

1 Opening of the meeting

The **Working Group on the Ecosystem Effects of Fishing Activities (WGECO)** met at ICES, Denmark from 8–15 April 2015. The list of participants and contact details are given in Annex 1. The chair, Anna Rindorf (Denmark) welcomed the participants and highlighted the variety of ToRs. The draft agenda was presented (Annex 2) and Terms of Reference for the meeting (see section 2) were discussed. Two additional topics were identified as being particularly in need for immediate advice and added to the work plan; risk based advice methods for sensitive species and the use of surveillance indicators in management. A plan of action was adopted with individuals providing presentations on particular issues and allocated separate tasks to begin work on all ToRs.

2 TORs for the 2015 meeting

The **Working Group on the Ecosystem Effects of Fishing Activities** (WGECO), chaired by Anna Rindorf, Denmark, will meet in ICES Headquarters, Copenhagen, Denmark 8–15 April 2015 to:

- a) Develop and assess indicators of Good Environmental Status of the benthic community;
- b) Develop indicators of scavengers, examine their relation to discard amounts and evaluate the potential effect of a landing obligation on the benthic ecosystem;
- c) Evaluate the ecological consequences of restoring stocks to MSY levels and the degree to which fisheries are “balanced;”
- d) Review, develop and assess indicators of Good Environmental Status of Distribution in the context of MSFD and management response;
- e) Identify data weaknesses in the data available to address WGECO ToRs and recommend priority areas for data collection and model development.

WGECO will report by 24 April to the attention of the Advisory Committee.

3 Develop and assess indicators of Good Environmental Status of the benthic community (ToR a)

ToR a “Develop and assess indicators of Good Environmental Status of the benthic community” was interpreted by WGECO as referring to indicators of GES for seafloor integrity. This interpretation includes the pressure acting on the benthic community. The most recent iteration of MSFD Descriptor 6: Seafloor integrity, i.e. the ICES recommendation to amend Decision 2010/477/EC, recommends the following GES criteria:

6.1. Damage to the seafloor, having regard to both pressure(s) on, and sensitivity of, habitats

- Extent of pressure(s) on the seafloor (single, multiple or cumulative) (6.1.1);
- Extent of the seafloor significantly affected by human activities for the different substratum types (including biogenic) (6.1.2).

6.2. Structural and functional condition of benthic community

- Presence of a particular species providing a key function (6.2.1);
- Multimetric indices assessing benthic community structure and function (6.2.2);
- Proportion of biomass or number of individuals in the macrobenthos above some specified length/size (6.2.3);
- Parameters describing the characteristics (shape, slope and intercept) of the size spectrum of the benthic community (6.2.4).

In DPSIR (Driver-Pressure-State-Impact-Response) terms these criteria reflect the Pressure (6.1.1), Impact (6.1.2) and several aspects of State (6.2.1-6.2.4). In the following sections, potential indicators for pressure (chapter 3.1.1) and two approaches aimed at linking impact to pressure: one based on trait-based vulnerability (chapter 3.1.2.1) the other on measured decreases in overall abundance of the benthic community (chapter 3.1.2.2) will be presented.

3.1 Damage to the seafloor, having regard to both pressure(s) on, and sensitivity of, habitats (criterion 6.1)

This criterion includes indicators which reflect the pressure on the seafloor for the extent of the area disturbed and the intensity at which this occurs, i.e. “Extent of pressure(s) on the seafloor (single, multiple or cumulative) (6.1.1)” as well as indicators which reflect the extent of the seafloor affected, i.e. “Extent of the seafloor significantly affected by human activities for the different substratum types (including biogenic) (6.1.2)” which essentially are based on the combination of the exposure to the pressure and the sensitivity of the habitat.

3.1.1 Extent of Pressure indicator (criterion 6.1.1)

The current indicators for the extent of fishing pressure on the seafloor are based on three indicators that were proposed to monitor the wider ecosystem effects of fishing (CEC, 2008):

- Distribution of fishing activities
- Aggregation of fishing activities

- Areas not affected by mobile bottom gears

These indicators were further developed by (Piet and Hintzen, 2012) and presented and discussed in WGECO (WGECO 2012). Although much progress has been made on further development and the consistent calculation of these indicators, their (sub)regional calculation required for the MSFD is still hampered by the fact that there are confidentiality issues that prevent access to the international VMS data. At present the EU FP7 BENTHIS project has created the first map of fishing intensity of (an) entire MSFD (sub)region(s): Figure 3.1 shows the Greater North sea and the Celtic seas. This figure illustrates that bottom trawling is patchy and that this patchiness needs to be taken into account to assess the impact of trawling on the benthic ecosystem (Rijnsdorp *et al.*, 1998);(Lee *et al.*, 2010a);(Ellis *et al.*, 2014). Combining this information with existing habitat maps (EUNIS level 3) allows estimates of fishing pressure per habitat. For example Figure 3.2 shows that at a spatial resolution of 1x1 minute (approximately 2 km²), 34.9% of the surface area of the sublittoral sandy habitat (EUNIS A5.2) is trawled at an intensity of less than once in every 10 years, whereas the area trawled between 1–2 and 2–5 times per year constitute 13.4% and 12.8% of the surface area of the habitat, respectively.

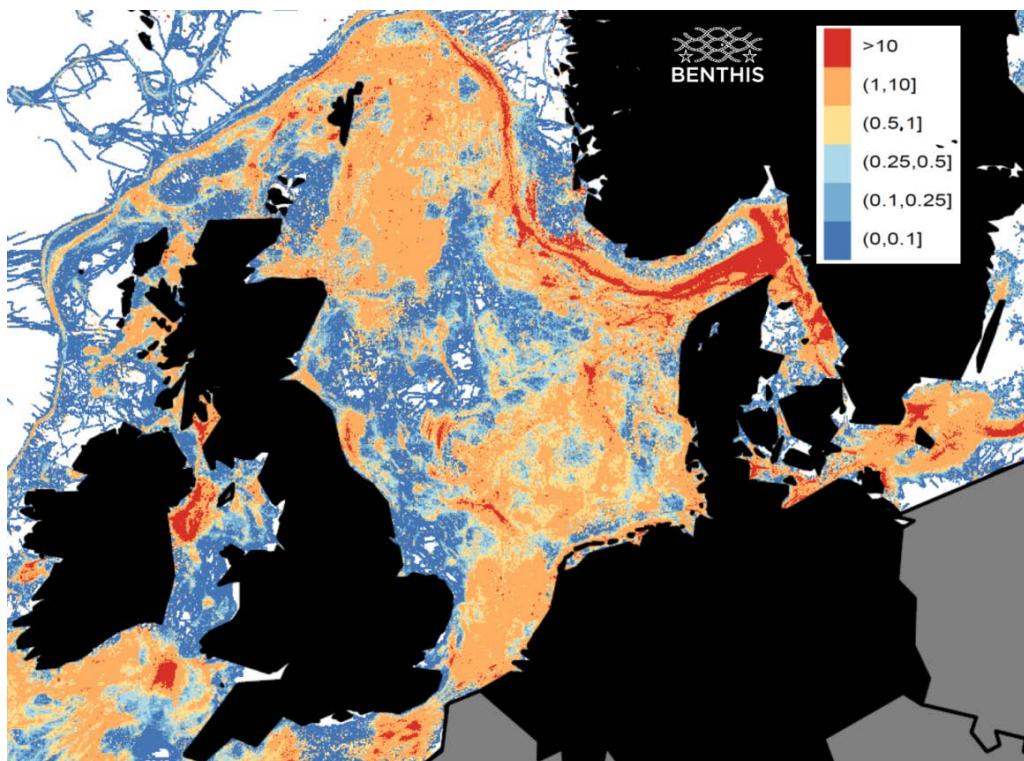


Figure 3.1. Bottom-trawl fishing intensity in the Greater North Sea and Celtic Seas over the period 2010–2012 expressed as the average number of times per year a unit area is fished. Note: VMS data only from countries in black. Source: BENTHIS

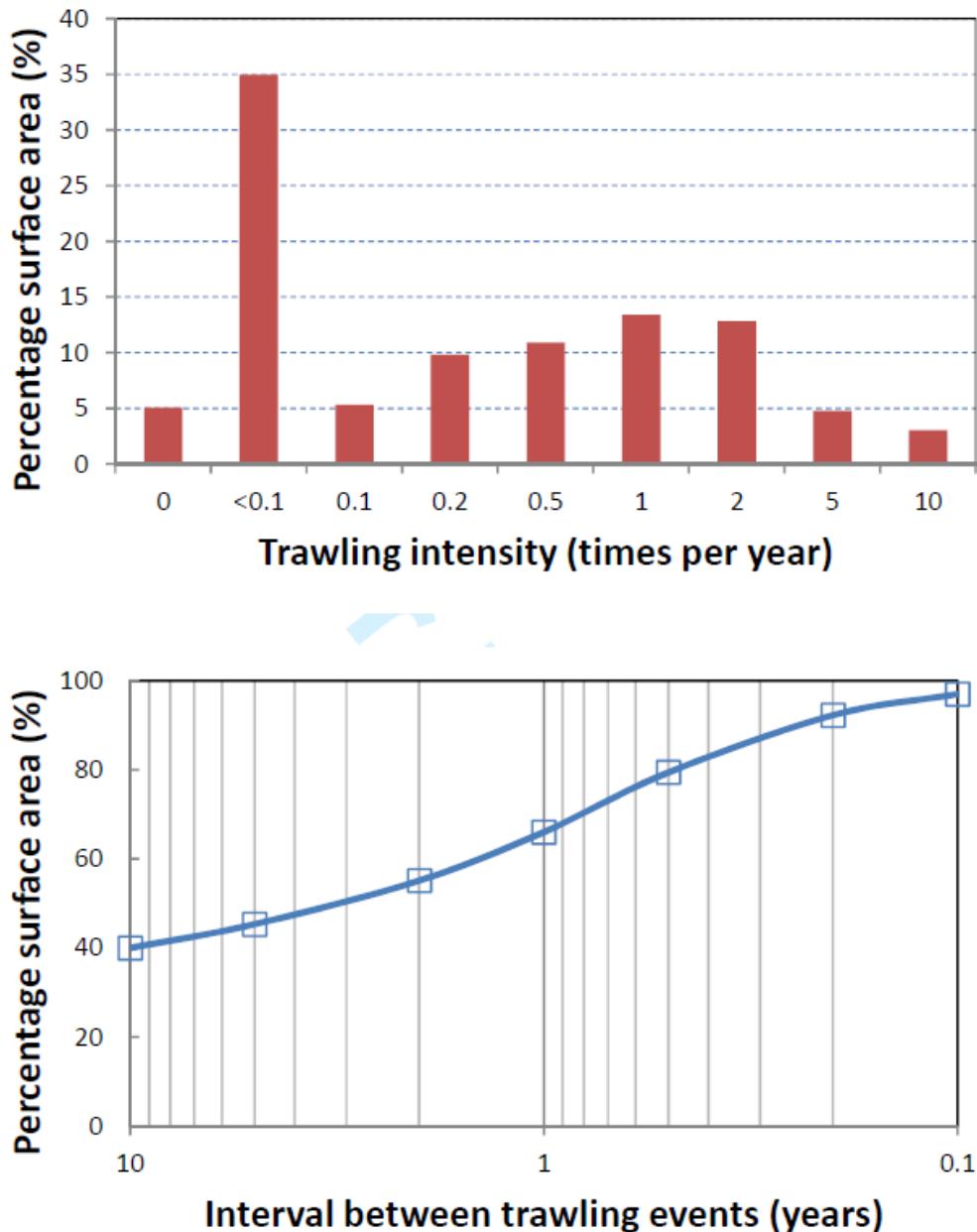


Figure 3.2. Upper panel (a): Percentage surface area of sublittoral sand (EUNIS habitat A5.2) trawled at different intensities. North Sea data 2010–2012 analysed at a resolution of 1x1 minute grid cells. Trawling intensities at the x -axis give the lower level of the bin. Lower panel (b): Cumulative surface area in relation to the time interval between two trawling impacts showing the surface area where benthos with a corresponding recovery time can recover and where this group of benthos are in the reference state. The time interval between trawling impacts was calculated as the reciprocal of the trawling intensities in upper panel.

The information contained in figure 3.2 can be condensed into a number of indicators of trawling pressure highlighting different aspects, e.g.

1. Proportion of the habitat that is not trawled (5.1%)
2. Proportion of the habitat that is trawled at least once in a year (34%)
3. Proportion of the habitat where 90% of the trawling effort is concentrated (54%)

The third pressure indicator gives an indication of the aggregation of fishing effort and estimates the proportion of the habitat encompassing the most intensively trawled grid cells that together represent 90% of the total fishing effort in the habitat. In this example, the area with 90% of the fishing effort is 54% of the surface area of the habitat which is well below the 95% of the surface area of the habitat that was trawled. Thus 41% of the habitat area could be protected by removing only 10% of the fishing effort.

3.1.2 Extent of Impact indicator (criterion 6.1.2)

When estimating these indicators based on the known pressure and the sensitivity of the benthic community on the seafloor (see section 9 on sensitivity) the term “significantly affected” needs to be defined as well as how this can be used to assess the status of seafloor integrity in relation to GES.

While the impacts of demersal fishing on the biological characteristics of the seabed have been well-studied, the approaches have tended to focus on assessing impacts on the structural (e.g. changes in species composition, diversity, etc.) characteristics of seabed biological assemblages. However, it is increasingly appreciated that considering changes solely in the structural attributes of benthic assemblages provides only limited capacity to inform on the implications for, arguably far more important, ecosystem function. The BENTHIS project (Bolam *et al.*, 2014, BENTHIS D3.4) therefore aims to bridge this knowledge gap by analysing data regarding the biological assemblages of a large number of stations covering a range of habitats across the European continental shelf. This was done using a biological traits analysis (BTA) in which the assemblages, and the differences between them, are quantified by their relative differences in the morphological, behavioural and life-history characteristics of their individuals, as opposed to their taxonomic differences. The BTA uses a suite of eleven traits that may determine the potential sensitivity of benthic taxa to trawling and their contribution to ecosystem functioning (Table 3.1). Each of these traits was subdivided into multiple ‘modalities’ chosen to encompass the range of possible attributes of all the taxa; for example, modalities for mobility were ‘swimming’, ‘burrowing’, ‘crawling’ and ‘sessile’. A total of 53 modalities among the eleven selected traits were identified (Table 3.2). This BTA approach affords the opportunity to understand the potential differences in ecological functioning due to fishing impacts, beyond that which would otherwise be possible from structural approaches.

Table 3.1 Biological traits and modalities used in analyses of changes in functional composition between different habitats types under varying level of fishing pressure in the EU project BEN-THIS.

Trait	Modalities
Morphology	Soft, Tunic, Exoskeleton, Crustose, Cushion, Stalked
Maximum body size (length) in mm	<10, 11–20, 21–100, 101–200, 201–500, >500
Longevity (maximum) year	<1, 1–<3, >3–10, >10
Larval development	Pelagic–Planktotrophic, Pelagic–Lecithotrophic, Benthic–Direct
Egg development location	Asexual/budding, Sexual-shed eggs-pelagic, Sexual-shed eggs-benthic, Sexual-brood eggs
Living habitat	Tube-dwelling, Burrow-dwelling, Free-living, Crevices/holes/under stones, Epi/endo-zoic/phytic, Attached
Sediment position	Surface, Infauna: 0–5 cm, Infauna: 6–10 cm, Infauna: >10 cm
Feeding mode	Suspension/filter-feeder, Surface deposit, Subsurface deposit, Scavenger/opportunist, Predator, Parasite
Mobility	Sessile, Swim, Crawl/creep/climb, Burrower
Bioturbators	Diffusive mixing, Surface deposition, Upward conveyor, Downward conveyor, None
Protection	No protection, Fragile, Tough skin or exoskeleton, Robust

In the following sections we first provide an example of how the information on the pressure (i.e. fishing) described in the previous section can be used to determine the vulnerability of (parts of) the benthic community and hence the potential impact of fishing. Next an example is provided of actual estimates of fishing impact, i.e. by reduction in overall abundance of the benthic community. This example will also show how estimates of fishing impact can be applied to assess the status of the benthic community in relation to GES.

3.1.2.1 Species composition affecting seafloor functioning

In a recent study by Rijnsdorp *et al.*, (submitted ICES Journal of Marine Science) the longevity trait was assumed to reflect the sensitivity of the benthic community to fishing disturbance as this can be considered a proxy for the recovery time. The part of the benthic community experiencing fishing disturbance with an intensity that results in an interval below their recovery time can be considered vulnerable to fishing. To what extent this vulnerability results in a measurable impact (i.e. reduced abundance) remains unclear as many other traits (e.g. Body size, Sediment position, Morphology, Protection) may determine the degree of impact.

Figure 3 shows the biomass distribution over the longevity classes as observed in high and low longevity habitats of the North Sea. The data were based on a large-scale assessment of benthic communities of the Greater North Sea (Bolam *et al.*, 2014). The biomass of both low and high longevity habitats have similar proportions of short lived taxa, but differ in the proportion of medium and long lived taxa (Figure 3.3).

Table 3.2. Traits and trait categories used during BENTHIS BTA. Where differences were adopted between infauna and epifauna, these are indicated in the third column. The abbreviation for each trait category as used for presentation within FCA plots and tables (see results) is given in brackets.

TRAIT	CATEGORIES (INFAUNA AND EPIFAUNA)	EPIFAUNAL CATEGORIES (WHERE DIFFERENT FROM INFAUNA)	TRAIT DEFINITION AND FUNCTIONAL SIGNIFICANCE
Size range (mm)	≤ 10 (s10)		
	11–20 (s11–20)		
	21–100 (s21–100)		
	101–200 (s101–200)		
	200–500 (s200–500)		Maximum recorded size of adult (as individuals or colonies). Implications for the movement of organic matter within the benthic system as large organisms hold organic matter (low turnover) within the system relative to small-bodied species (high turnover) (Pearson and Rosenberg, 1978).
Morphology	>500 (s500)		
	Soft (mSoft)	Round-bodied (mRound)	External characteristics of the taxon. For the infauna, mSoft are represented mainly by annelid worms, mTunic by tunicates, mExo represents chitinous (lower crustaceans) and calcareous-shelled (e.g. bivalve and gastropod molluscs, echinoderms, higher crustaceans). Crustose, cushion and stalked traits are shown by various sponges, hydroids and bryozoans. For epifaunal traits, mRound is represented by all taxa not showing the other trait categories and is represented by a wide range of non-colonial taxa such as crustaceans, molluscs, annelids and echinoderms.
	Tunic (mTunic)	Flat/encrusting (mFlat)	
	Exoskeleton (mExo)	Stalked/pen-shaped (mPen)	
	Crustose (mCrus)	Stalked/fan-shaped (mFan)	
	Cushion (mCush)	Stalked/complex (mComp)	
	Stalked (mStal)		
Longevity (years)	<1 (l1)		
	1–2 (l1to2)		
	3–10 (l3to10)		
	>10 (l10)		Maximum reported lifespan of the adult stage. Indicates the relative investment of energy in somatic rather than reproductive growth and the relative age of sexual maturity, i.e. a proxy for relative r- and k- strategy (Pearson and Rosenberg, 1978). Short-lived taxa (l1) include ostracods and small amphipods, while the molluscs Buccinum and Arctica represent some of the long-lived taxa.
Larval development strategy	Planktotrophic (ldPk) Lecithotrophic (ldLc) Direct (ldD)		Indicates the potential for dispersal of the larval stage prior to settlement from direct (no larval stage, e.g. cumaceans, tanaids), lecithotrophic (larvae with yolk sac, pelagic for short periods, e.g. terebellid worms) to planktotrophic (larvae feed and grow in water column, generally pelagic for several weeks, e.g. sponges, cnidarians). Affects ability to recover from disturbance with planktonic recruitment affording potentially faster recolonization than lecithotrophic and direct development (Thrush and Whitlatch, 2001).

TRAIT	CATEGORIES (INFAUNA AND EPIFAUNA)	EPIFAUNAL CATEGORIES (WHERE DIFFERENT FROM INFAUNA)	TRAIT DEFINITION AND FUNCTIONAL SIGNIFICANCE
Egg development location	Asexual/fragmentation (edAsex) Eggs – pelagic (edSex_pel) Eggs – benthic (edSex_ben) Eggs – brooded (edSex_br)		Indicates dispersal via the egg stage and the potential susceptibility of eggs to damage from fishing. Benthic eggs (e.g. some eunicid worms) are generally more concentrated over smaller areas than eggs released into the pelagia (e.g. hesionid worms). Asexual reproduction allows the potential to increase numbers rapidly, particularly following disturbance. Brooding is widespread within the lower crustaceans (e.g. amphipods).
Living habit	Tube-dwelling (lhTube) Burrow-dwelling (lhBurrow) Free living (lhFree) Crevice/under stone (lhCrev) Epi/endo zoic/phytic (lhEpi) Attached to bed (lhAtt)		Indicates potential for the adult stage to evade, or to be exposed to, physical disturbance.). Various lhTube (e.g. serpulid worms), lhBurrow (some bivalve molluscs), lhCrevice (such as piddocks), lhFree (e.g. eumalacostracan crustaceans), lhEpi (e.g. bryozoans) and lhAtt (e.g. ascidians, bryozoans) taxa will vary in their acute responses to trawling depending on this trait (in combination with those of other traits such as mobility and sediment position).
Sediment position	Surface (spSurf) 0–5 cm (spInf_0to5) 5–10 cm (spInf_6to10) >10 cm (spInf_10)		Typical living position in sediment profile. Organisms occupying surficial (e.g. mytilid molluscs, sponges) or shallow positions in the sediment (some bivalves) are more likely to contact trawl gear than those living deeper (e.g. some worms). Sediment position also has implications for the effect of the organism to affect sediment-water nutrient and/or oxygen exchange.
Feeding mode	Suspension (fSusp) Surface deposit (fSurf) Subsurface deposit (fSub) Scavenger (fScav) Predator (fPred)		Feeding mode has important implications for the potential for transfer of carbon between the sediment and water and within the sediment matrix. Feeding mode also has important repercussions for many biogeochemical processes (Rosenberg, 1995).
Mobility	Sessile (mobSess) Swim (mobSwim) Burrow (mobBur) Crawl (mobCrawl)		Adults of faster moving species are more likely to evade capture by trawl gear than slow-moving or sessile individuals. Mobility also affects the ability for adult recolonization of disturbed areas.
Bioturbation	Diffusive mixer (bDiff) Surface deposition (bSurf) Upward conveyor (bUpward) Downw. conv.(bDownward) None (bNone)		Describes the ability of the organism to rework the sediments. Can either be upward (e.g. maldanid worms), downward (e.g. oweniid worms), onto the sediment (many suspension-feeders) or mixing of the sedimentary matrix (e.g. glyceriid worms). Bioturbation mode has important implications for sediment-water exchange and sediment biogeochemical properties.

TRAIT	CATEGORIES (INFAUNA AND EPIFAUNA)	EPIFAUNAL CATEGORIES (WHERE DIFFERENT FROM INFAUNA)	TRAIT DEFINITION AND FUNCTIONAL SIGNIFICANCE
Protection		Fragile (pFrag) Unprotected (pUnprot) Protected (skin/exoskeleton) (pExo) Protect (robust shell) (pRob)	Describes the capacity to withstand physical disturbance and thus the potential for the adult population to remain viable following acute fishing. Examples include; pFrag (<i>Atrina</i> sp.); pUnprot (<i>Capitella</i> sp.); pExo (<i>Bathyporeia</i> sp., <i>Cancer</i> sp.); and pRob (<i>Arctica</i> sp.).
Bed/reef formers		None (brNone) Reef-builder (brReef) Bed-former (brBed)	Important for affecting a number of ecological properties such as biodiversity, productivity and sediment stability. Reef-builders (e.g. <i>Sabellaria</i> sp.) create an elevated structure on the seabed through chemical precipitation or concretions while bed-formers (e.g. mussels) form dense aggregations that visually dominate the seabed.

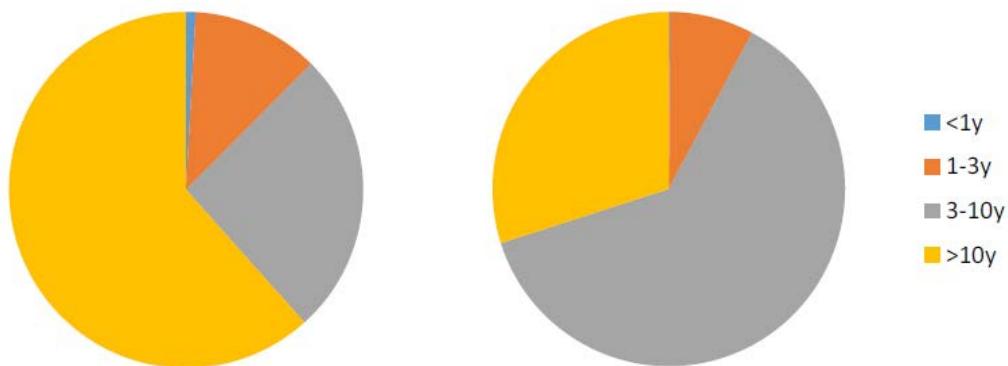


Figure 3.3. Biomass distribution of various longevity groups of the benthos in two habitats of which the benthic community composition differs in longevity: high longevity (left) and low longevity (right) habitats for the Greater North Sea. Longevity reflects the maximum reported lifespan of the adult stage. Data derived from Bolam *et al.*, (2014).

Applying the same trawling intensity distribution shown in Figure 3.2 to both high and low longevity habitat, the indicator showing 34% of the area is trawled less than once per year indicates that 56% in the low longevity habitat and slightly more in the high longevity habitat because this also contains some benthos with longevity <1 year. The higher vulnerability of the benthic community and thus larger trawling impact in the high longevity habitat is notably related to the greater proportion of biomass in the highest longevity class.

The high and low longevity habitats have different benthic communities regarding their functioning (Figure 3.4). The biomass of the high longevity habitat is dominated by surface depositors. Surface deposition here refers to the bioturbation mode and refers to organisms which take food from the overlying water column (either *via* suspension or filter feeding) and place their faeces onto the sediment. The surface depositors and the other bioturbators have different longevity distributions (Figure 3.4). Surface depositors are mainly long lived (>10 years), whereas the bioturbators have an intermediate longevity (3–10 years). Applying the trawling intensity distribution of Figure 3.2 to both functional groups shows that the surface area where biodepositors are not at risk is reduced to 57.3% (high longevity) and 57.6% (low longevity) while for the bioturbators this is 50.6% (high longevity) and 52.4% (low longevity).

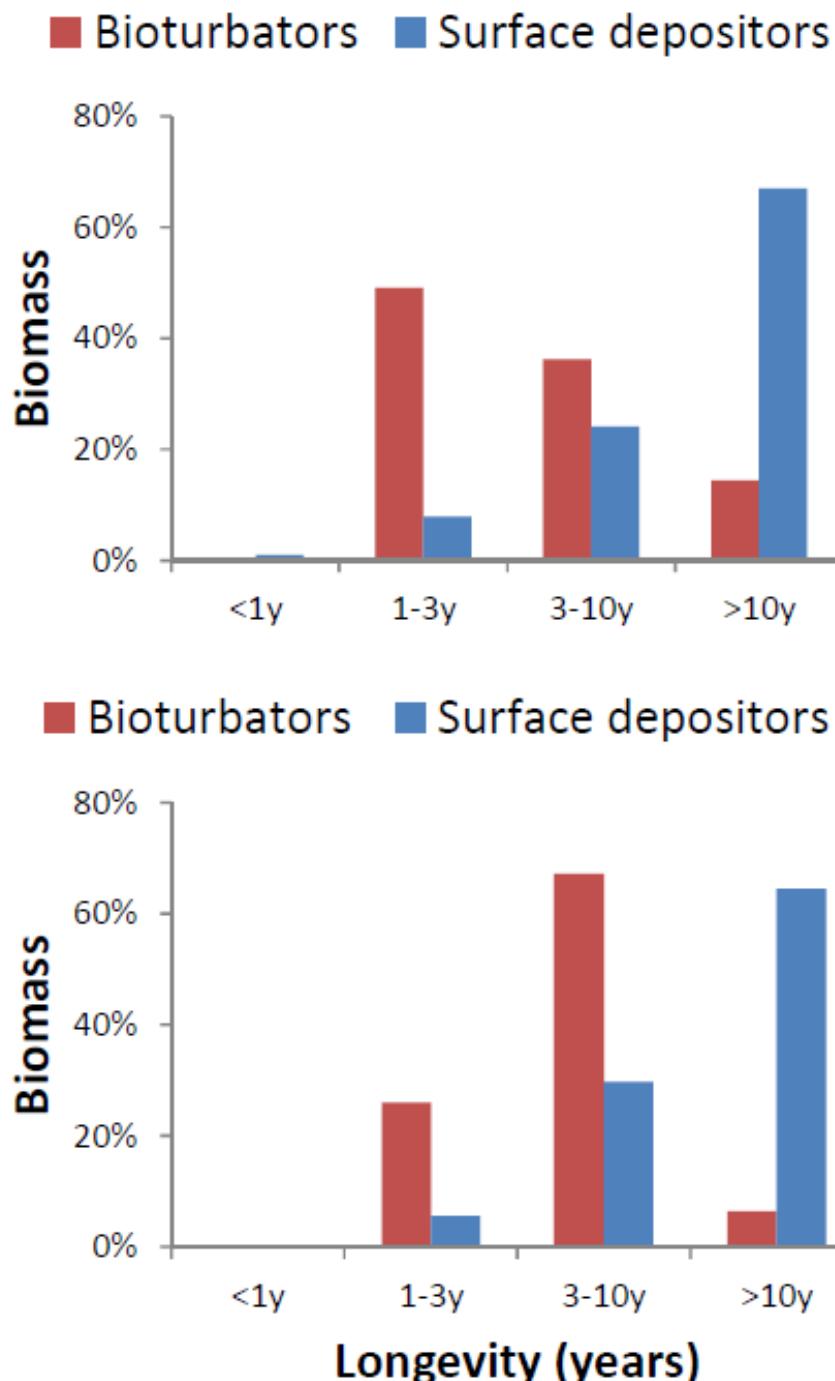


Figure 3.4. Biomass distribution over longevity classes for two functional groups (surface depositors and bioturbators) in High longevity habitat (top panel) and Low longevity habitat (bottom panel). Data Greater North Sea from Bolam *et al.*, (2014).

The use of this framework on real data shows how different types of habitats (low and high longevity) have communities with a different longevity composition and as such they differ in their vulnerability to trawling impact. The results furthermore show how different functional groups within these habitats have different longevity compositions; surface depositors, likely to be predominantly bivalves, are longer living and hence more vulnerable to trawl impact than biodepositors. Such findings indicate how trawling can change the ecological function of an area (e.g. Tillin *et al.*, 2006; de Juan *et al.*, 2007). The results presented here show limited differences be-

tween habitats (~1% difference) and functional groups (~10% difference), presumably because data are aggregated over large spatial scales and mixed sediments (Bolam *et al.*, 2014). Larger differences in benthic community structure (e.g. longevity composition) and ecological function have been found between habitats sampled at smaller spatial scales (van Denderen *et al.*, submitted) and such datasets can be used to construct a proof of principle for the ecological part of the assessment framework.

3.1.2.2 Overall abundance

This example will show how estimates of fishing impact can be applied to assess the status of the benthic community in relation to GES. For this we need to distinguish two different spatial scales:

- Spatial unit (i.e. 100x100 m)
- Habitat (i.e. EUNIS level 3)

Seafloor is considered significantly impacted if, for the particular aspect considered, the mortality rate > rate of recovery which depends both on the instantaneous effect of the pressure on the benthic community in a specific spatial unit of a habitat (a in Figure 3.5), the time to recovery (c in Figure 3.5) and the trawling intensity. The sensitivity of the habitat is reflected in both the instantaneous effect and the time to recovery.

Sciberras and Hiddink (2014) used systematic review methodology (Pullin and Stewart 2006; Higgins and Green, 2008) and meta-analysis (Arnqvist and Wooster, 1995; Gurevitch and Hedges, 1999) to examine the magnitude of the response of benthic invertebrates in different habitats (Figure 3.6) and different type of bottom fishing gear. A comprehensive search of peer-reviewed scientific literature and grey literature (up until February 2014) was conducted to compile a database of studies that documented and compared the biological effects of experimentally fished areas with bottom mobile fishing gear (the impacted area) to non-fished areas (the control area) (Hughes *et al.*, 2014). Non-fished areas were generally areas of similar environmental characteristics to the fished areas that have not been fished for years, for example because of the presence of marine protected areas or area closures to mobile bottom fishing gear, or areas that were only very lightly fished.

The natural logarithm transformed response ratio, LnRR (Hedges *et al.*, 1999) was used as the effect size, which is better suited than other metrics for a study of changes brought about by an impact because it is designed to quantify the proportionate change that results from the intervention (Goldberg *et al.*, 1999; Hedges *et al.*, 1999). LnRR was used instead of response ratio (RR) because it linearizes the metric so that changes in the denominator and numerator are treated equally and yields better sampling distributions (Hedges *et al.*, 1999). The response ratio is defined as the ratio of the mean abundance estimate measured inside (impact) and outside (control) the fished area for a control-impact design study or before (control) and after (impact) for a before-after study (Hedges *et al.*, 1999):

$$LnRR = \ln \left(\frac{\bar{X}_{Impact}}{\bar{X}_{Control}} \right)$$

A higher fishing intensity in a sensitive habitat results in a higher degree of impact and hence lower seafloor integrity (see Figures 3.7 and 3.8). While this approach allows an estimation of the seafloor integrity in each spatial unit of a particular habitat it is still unclear where the threshold lies for a “significant impact”, at the scale of the spatial unit (i.e. percentage relative to undisturbed) or what the scientific basis would

be to set a GES boundary for the proportion of the surface area not significantly impacted, at the scale of the habitat.

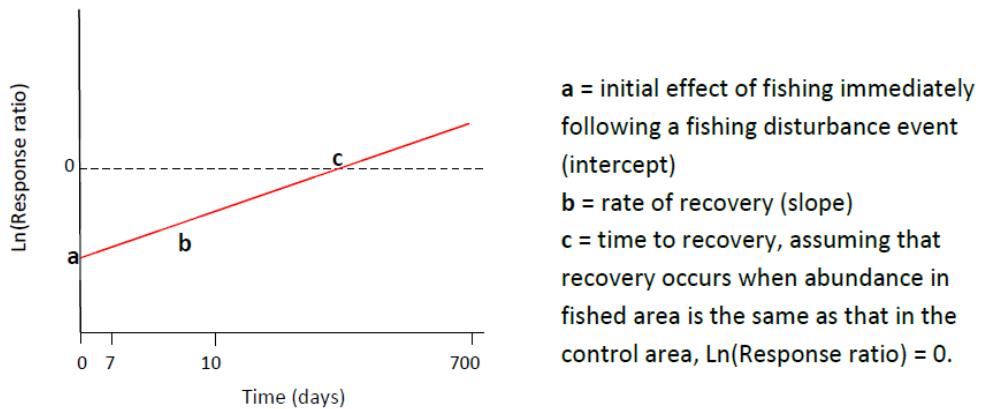


Figure 3.5. A diagrammatic representation of the response to fishing (ln-transformed response ratio) with time since the last fishing disturbance event (in days). From Sciberras and Hiddink (2014).

Based on the linear regression given in Figure 3.6 we can construct the Figure 3.7 while Figure 3.8 takes some of the uncertainty in the linear regressions into account by using the lower 95% confidence limits of the intercept. These figures quantify the seafloor integrity (relative to undisturbed) per spatial unit within a habitat depending on the (historic) trawling intensity.

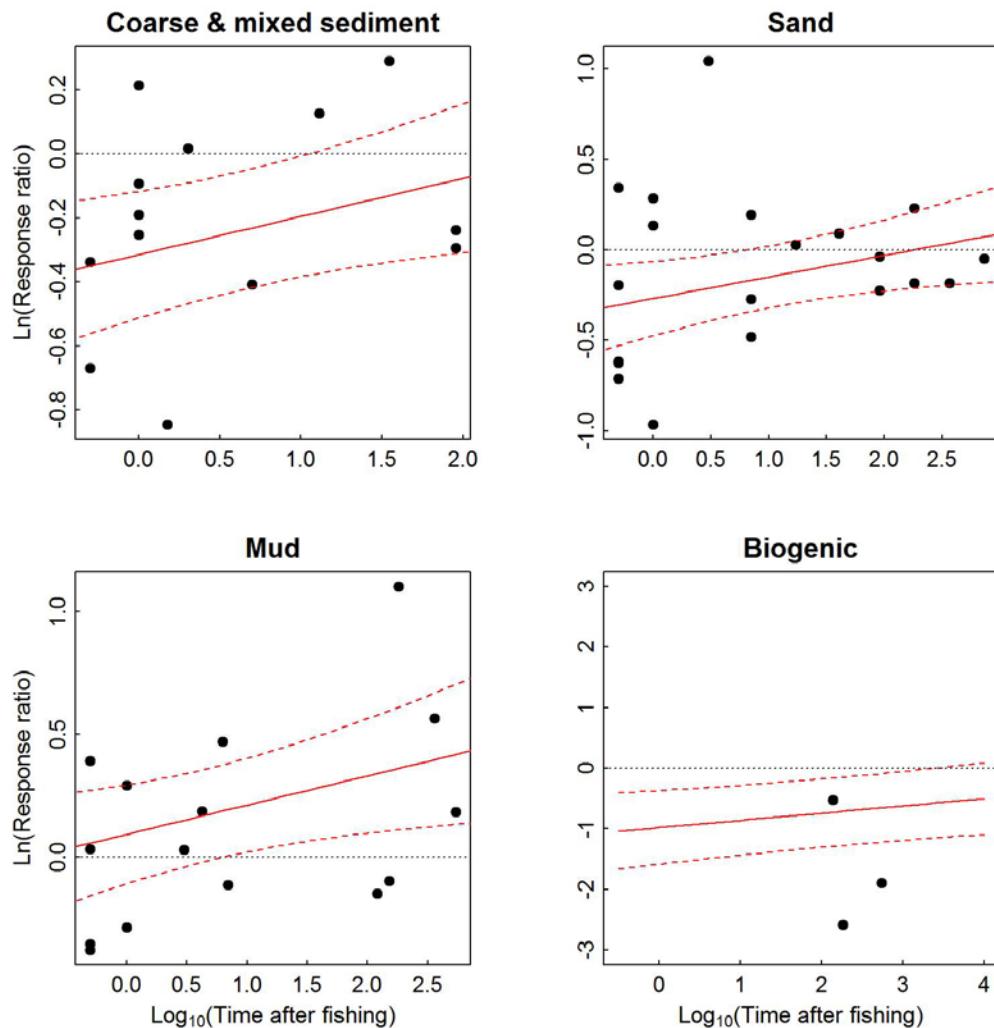


Figure 3.6. The change in response of total benthic community abundance with time following a fishing event, for benthos occurring in coarse and mixed sediment, sand, mud and in biogenic substrata. The vertical dotted line at ($\ln\text{RR}$) = 0 represents equal abundance in fished and control area. The fitted model and 95% confidence intervals are plotted in red (Fitted model for coarse and mixed sediment: $y = -0.31 + 0.12\log_{10}\text{time}$; fitted model for sandy sediment: $y = 0.26 + 0.12\log_{10}\text{time}$; fitted model for muddy sediment: $y = 0.1 + 0.12\log_{10}\text{time}$; fitted model for biogenic habitats: $y = -0.98 + 0.12\log_{10}\text{time}$). Time unit is in days. From Sciberras and Hiddink (2014).

The Figure 3.7 shows that if a 5% decrease from undisturbed is allowed before it is considered “significantly impacted” only spatial units in coarse and mixed sediment trawled $>0.2 \text{ yr}^{-1}$ are considered affected. For biogenic substratum this is $>0.1 \text{ yr}^{-1}$. Mud is not shown as the Figure 3.7 suggests no significant impact.

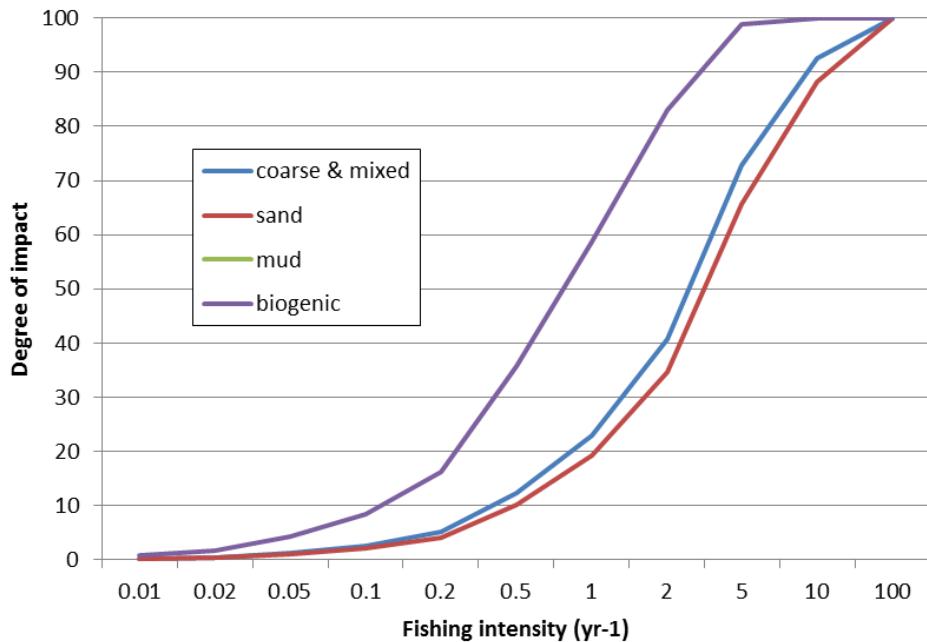


Figure 3.7. Seafloor integrity depending on fishing intensity and sensitivity of the habitat (based on the fitted model given in figure 3.6). Mud is not shown as the figure 3.8 suggests no significant impact.

The more conservative Figure 3.8 (i.e. based on lower bounds shown in Figure 3.6) shows that if a 5% decrease from undisturbed is allowed before it is considered “significantly impacted” only spatial units in mud trawled $>1 \text{ yr}^{-1}$, coarse and mixed sediment trawled $>0.1 \text{ yr}^{-1}$ and biogenic substratum trawled $>0.01 \text{ yr}^{-1}$ are considered affected. Aggregation across the spatial units within a habitat then allows an estimation of the proportion of a habitat that is “significantly impacted”. There is currently no scientific basis that allows the setting of a boundary for the proportion of the area of a habitat than can be “significantly impacted” before the habitat is considered not to be in GES. This appears to be a societal as much as a scientific decision.

This method and consequential estimate of the area “significantly impacted” as well as the setting of the GES boundaries depend on the spatial and temporal scale on which the pressure data are aggregated (Piet and Quirijns, 2009).

Here we used a spatial scale of about 1 minute latitude by $\times 1$ minute longitude for pragmatic reasons because (1) beyond this the rounding of GPS position in VMS may cause a bias in the analyses (Rijnsdorp *et al.*, submitted) (2) trawling was shown to be sufficiently randomly distributed at this level of resolution (Rijnsdorp *et al.*, 1998; Ellis *et al.*, 2014) and it was considered to be appropriate in other studies (Lee *et al.*, 2010b); (Gerritsen *et al.*, 2013).

For the temporal scale we used year because this is commonly used in fisheries management. As with the spatial scale this implicitly assumes the pressure is distributed randomly within this unit. If there are solid reasons to suspect this is not the case (e.g. a seasonal fishery only operating in a specific time of year) an alternative, more appropriate temporal scale can be selected.

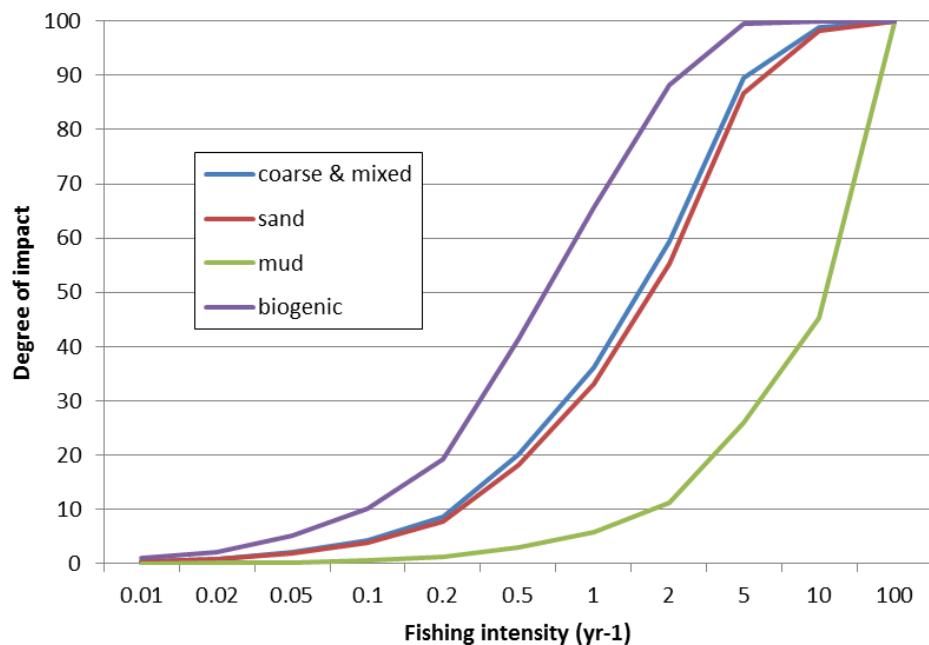


Figure 3.8. Seafloor integrity depending on fishing intensity and sensitivity of the habitat (based on the lower bound of the intercept in Figure 3.6).

3.2 Structural and functional condition of benthic community (criterion 6.2)

This criterion reflects several aspects of state which not only involve structural aspects, either in species-composition or size-distribution, but also the functioning of the benthic community, based on single species, i.e. “Presence of a particular species providing a key function (6.2.1)”, as well as the entire community as reflected by the “Multimetric indices assessing benthic community structure and function (6.2.2)”. The “Proportion of biomass or number of individuals in the macro benthos above some specified length/size (6.2.3)” and “Parameters describing the characteristics (shape, slope and intercept) of the size spectrum of the benthic community (6.2.4)” both reflect the size distribution.

3.2.1 Single-species indicators providing a key function (criterion 6.2.1)

Within HELCOM there is a process where specific key species are identified as important bioturbators. WGECO was not aware of any other initiatives.

3.2.2 Multimetric indices (criterion 6.2.2)

3.2.2.1 Linking biological traits of benthic species with ecosystem functions

The biological traits of benthic species can be used to characterize the benthic community through its sensitivity to different pressures (including fishery) as well as its functioning and the relationship between the two. Benthic ecosystems in offshore areas provide a number of important functions in marine environments. These include bioturbation/biodeposition processes (Pearson, 2001; Biles *et al.*, 2002; Mermilliod-Blondin, 2011) involving particle reworking (Kristensen *et al.*, 2012) and the oxygenation of sediments (Kristensen and Kostka, 2005) through burrow ventilation (Kristensen *et al.*, 2012); prey supply (Bruno and O’Connor, 2005) and the special case

of benthic-pelagic coupling (Gili and Coma, 1998; Iken *et al.*, 2001); and carbon/silicon sequestration (Maldonado *et al.*, 2012).

Linking changes in species composition to alterations in these functions can be challenging as there are so many species of marine invertebrates and most are poorly studied. Therefore it can be difficult to predict the consequence of species changes to ecosystem functioning. Recently, biological traits expressed by organisms have been used to assess how ecosystem functioning varies with changes to species composition (Bremner *et al.*, 2003). The important bioturbation/biodeposition functions of benthic invertebrates vary strongly according to their biological traits (e.g. mode of feeding, mode of locomotion, burrowing depth; Pearson, 2001; Solan *et al.*, 2004; Mermilliod-Blondin 2011), and recently Queirós *et al.*, (2013) provided a traits-based classification of 1033 benthic invertebrate species from the northwest European continental shelf. They further combined individual traits to produce a single community-level metric, Bioturbation Potential (BPc), to quantify this ecosystem function. Tubes and burrows are special cases of bioturbation as they have a direct opening to the surface and thereby influence the oxygen concentration and oxygen penetration depth in marine sediments. Burrows are known to enhance total microbial metabolism by stimulating respiration and biological traits have also been associated with this function (Kristensen and Kostka, 2005).

Recently, the EU FP7 project BENTHIS created a table of biological traits and modalities for use in biological traits analysis (Tables 3.1 and 3.2). BENTHIS examined those traits to see how well they measured the above aspects of ecosystem functioning (Table 3.3) and considered whether a change to the mix of modalities within the trait could affect each function. That assessment was based on expert opinion and the scientific literature. It was found that all of the functions that we identified could be measured by one or more of the traits identified by BENTHIS. Two traits, Size Range (mm) and Sediment Position (Table 3.1) could respond to 4 of the 5 functions, while 5 of the 12 traits did not map directly to any of the ecosystem functions.

3.2.3 Linking biological traits of benthic species with pressures

Having identified a set of traits that could be used to measure changes in ecosystem functioning, WGECO then considered how those same traits responded to the 8 general pressures outlined in the MSFD (Table 3.4). Again, expert opinion and the scientific literature was used to populate the table (Table 3.5). Mobility was deemed an indicator for 7 of the 8 general pressures, while Living Habitat and Size responded to 6 and 5 pressures respectively. Only Longevity did not directly match with a specific pressure (Table 3.5). Sensitivity to all pressures, except 'Other physical disturbance (underwater noise, marine litter, electromagnetic changes)' was affected by at least one of the traits, and sensitivity to Physical Damage could be directly assessed using 10 of the 13 traits.

3.2.4 Relationship between pressure and function

The human pressures identified in the MSFD (Table 3.4) are not all expected to impact the functioning of benthic ecosystems in the same way. For example, pressures affecting water quality (e.g. contamination by synthetic compounds; enrichment of N and P) will affect different species than pressures which cause physical damage through abrasion or which cause death or injury by collision. Therefore it is important that each pressure is considered separately. This applies both to analyses of species and of traits matrices.

To that end we propose a mechanistic approach that disentangles (1) the effects of the pressure on the benthic community which is determined by the sensitivity of the benthos to that pressure (Table 3.4) and (2) how the affected benthic community, with a different species-composition will display a change in its functioning (Table 3.5).

Table 3.3. The list of pressures identified in the Marine Strategy Framework Directive.

Pressure specific	Pressure general
Smothering	Physical loss
Substrate Loss	
Changes in siltation	
Abrasion	Physical damage
Selective Extraction (non-living) resources	
Death or injury by collision	
Underwater noise	
Marine Litter	Other physical disturbance
Electromagnetic changes	
Emergence regime change	
pH changes	
Water flow rate changes	Interference with hydrological processes
Thermal regime changes	
Salinity regime changes	
Change in wave exposure	
Synthetic compounds	
Non-synthetic compounds	Contamination
Radionuclides	
Nitrogen and Phosphorus enrichment	Enrichment
Input of organic matter	
Microbial pathogens	
Non-indigenous species	Biological disturbance
Selective extraction of species	
Barrier to species movement	Other physical disturbance

Table 3.4. Biological traits of benthic marine invertebrate species which may respond to the pressures identified in the Marine Directive (Table 3.2).

		Pressure									
Trait		Physical loss	Physical dam age	Other physica l disturbance	Interference with hydrolo gical process es	Contami nation	Enrich ment	Biologi cal disturbance	Barrier to specie s move ment	Total Pressu res Meas ured by Trait	
Size range (mm)		X		X		X	X	X	X	5	
Morphology		X								1	
Morphology (Epifaun a)		X								1	
Longevity (years)										0	
Larval development strategy								X	X	1	
Egg development location						[X]			X	2	
Living habit	X	X		X	X		X	X	X	6	
Sediment position	X	X		X	X			X	X	5	
Feeding mode		X				X	X	X		4	
Mobility	X	X		X	X	X	X	X	X	7	
Protection		X				X					2
Bed/reef formers		X		X		X					3
Bioturbation	X	X		X	X						4
Total Traits Responding to Pressure	4	10	0	6	8	3	4	6			

Table 3.5. Biological traits of benthic marine invertebrate species which may influence benthic ecosystem functions.

Trait	Function					Total No. Functions Measured by Trait
	Bioturbation/Biodeposition		Foodweb		Storage	
	Particle Reworking	Oxygenation of Sediments	Predator Prey	Benthic-Pelagic Coupling	Carbon/Silicon Sequestration	
Size range (mm)	X	X	X		X	4
Morphology			X			1
Morphology (Epifauna)						0
Longevity (years)						0
Larval development strategy						0
Egg development location						0
Living habit	X	X	X			3
Sediment position	X	X	X	X		4
Feeding mode			X	X		2
Mobility	X	X	X			3
Protection						0
Bed/reef formers				X	X	2
No. Traits Potentially Responsive to Function	4	4	6	3	2	

3.2.5 Size based metrics (6.2.3 and 6.2.4)

Community body size largely determines the types and strengths of energy flows through an ecosystem. In particular, biomass, production, oxygenation of sediments, and predator-prey interactions respond to changes to ecosystem size structure. Changes in community size composition may therefore have consequences for ecosystem functioning. Conversely, invertebrate growth can be determinant or indeterminant and follow exponential, power or sigmoidal growth functions (Hirst and Foster, 2013). Therefore it can be difficult to introduce size into the type of species-composition-based metrics that can be derived using the traits considered for e.g. the multimetric indices. The type of size-based metrics that are required here can only be calculated if size is actually measured, at least for a suite of key species for which size is of relevance for their functioning.

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

4.1 Approach

In 2014, WGECO recommended priority areas of study to determine the ecosystem consequences of landing obligations (Borges *et al.*, 2014; ICES, 2014). Extracting additional biomass in the form of otherwise discarded organisms may have consequences for the populations from which they are extracted, scavenger species that feed on discarded organisms and secondary responses in the ecosystem. The landing obligation is expected to decrease the amount of discards considerably. A direct consequence of banning discards may therefore be the potential creation of a food shortage for scavenging species.

The effect of this shortage depends on the ability of the scavengers to compensate by switching to other food sources and on the changes in conversion efficiency of their food. This may limit the direct effects on these species, but may also cause unpredictable cascading effects on other species through increased predation and/or competition. The effect of a reduction in food for seabirds might be expected to lead to decreased populations of the species most dependent on discards such as large generalist seabird species (Bicknell *et al.*, 2013). Besides seabirds, a wide range of other species was identified to feed on discards, from marine mammals to benthos (Was-senberg and Hill, 1990; Svane *et al.*, 2008). Knowledge is insufficient to determine larger scale effects. Several aspects were only addressed in a limited number of studies:

- The relative contribution of discards to the food consumption of the large-scale scavenger community is unknown as is the conversion efficiency of the ingested matter to scavenger biomass;
- The extent to which the scavenging community will be able to compensate by feeding on alternative prey is unclear.

WGECO therefore adopted the following Term of Reference in 2015:

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

This ToR will be addressed over a two-year work plan. This year's report focuses on the identification of the key scavenging community and discusses the issues to be accounted for when developing indicators of scavengers and identifying their relationship with the discarded amounts.

Discarding within EU fisheries has previously been a requirement under the EU common fisheries policy. With the introduction of the landing obligation and the banning of discards some scavenger species which are reliant on discards as a source of carrion may be affected. Therefore, indicators of scavengers need to be developed to evaluate the effect of the management change on the ecosystem. The development and testing of indicators for key scavengers will be addressed in a case study in next year's meeting.

4.2 Identify the key scavengers

4.2.1 What is a scavenger?

Marine scavengers are defined by Britton and Morton (1994) as organisms which are “able to detect carrion, usually by either distance or touch chemoreception, or both, deliberating to move toward it, and eventually consume either part or all of it”.

Bengtson, 2002 suggests that “*Scavenging and predation are often two sides of the same behaviour, and detritus-feeders are bound to engulf countless living microbes. Most organisms are not confined to a single mode of life, so the same organism may be predator, scavenger, parasite, etc.—and, of course, prey. Phenomena in nature tend to have fuzzy edges, and terminology should not lead us to forget that.*” Further “fuzzy edges” exist between scavengers and filter-feeders, and Walker and Bambach (1974) point out that scavenging is not a sharply defined feeding category, but merges with that of deposit-feeders. Filter-feeders could thus also be described as scavengers.

A facultative scavenger can be defined as an animal “*that scavenges at variable rates but that can subsist on other food resources in the absence of carrion*”, while an obligate scavenger can be defined as an animal “*that relies entirely or near entirely on carrion as food resource*” (Moleon *et al.*, 2014).

Britton and Morton (1994) indicate that many marine animal phyla include scavengers; Turbellaria; Nemertea; Nematoda; Polychaeta; Mollusca; Arthropoda; Echinodermata; Fish; Seabirds, and mammals. Probably very few of these could be described as “obligate scavengers”, indeed Britton and Morton (1994) suggest that they could not exist in the marine environment due the paucity of material to forage upon, although they go on to say that “*if there are obligate scavengers among marine animals, they will most likely be found among the Crustacea and the Gastropoda*”. Their focus was on lysianassoid Amphipoda and nassariid Gastropoda.

This absence of obligate scavengers is contested by Kaiser and Moore (1999) who suggest that the lysianassoid amphipod *Orchomene nanus* is a good candidate for an “obligate scavenger”, and one that makes use of discarded fish. Ruxton and Houston (2004) also demonstrated theoretically that obligate scavengers could exist in marine environments. The scavenging isopod *Natatolana borealis* would be another possible candidate (Wong and Moore, 1995).

Among fish species, the most obvious candidate would be the hagfish. In a baited camera study Martinez *et al.*, (2011) found that hagfish (*Myxine glutinosa*) was the most abundant species attracted to bait. It should be noted however, that the other most common species were flatfish (mainly dabs *Limanda limanda*), whiting (*Merlangius merlangus*) and haddock (*Melanogrammus aeglefinus*) emphasizing the continuum between predator and scavenger. In this context, even hagfish have been shown to occasionally act as predators (Zintzen *et al.*, 2011).

Finally, Beasley *et al.*, 2012, suggest the seafloor environment may have encouraged the evolution of a more diverse range of facultative scavengers that make more use of carrion in their diet than occurs in terrestrial ecosystems.

In addressing this ToR we therefore recognize that “scavengers” actually exist on a continuum between those that are close to obligate scavengers through to predators that will occasionally scavenge. The animals that will be most affected by a landing obligation, and hence removal of discards, are likely to be those towards the obligate end of this continuum.

4.2.2 Identification of key scavengers

Key scavengers were identified from field studies which investigated the aggregation of organisms after presenting discards to them as bait. Scavenging organisms included species which were able to detect discards and move towards them for consumption. The key species were consequently mainly identified based on their numerical dominance and only indirectly based on their dependence on discards. Studies that investigated the aggregation or increased abundances of scavengers after a trawl passage were excluded as damaged organisms may alter scavenger's attraction (Jenkins *et al.*, 2004). Furthermore, studies documenting catch compositions from baited fisheries could also be used herein to document scavenger species, as these species are also attracted to dead carrion on the seabed. In 2015, WGECO focused on studies which are not commercial baited fisheries as these will be addressed next year (landings and discards from commercial fisheries with baited lines, pots and traps).

7 studies in the southern hemisphere, Antarctica, Australia and South Pacific (Wassenberg and Hill, 1987; Wassenberg and Hill, 1990; Collins *et al.*, 1999; Hill and Wassenberg, 2000; Morton and Yuen, 2000; Smale *et al.*, 2007; Svane *et al.*, 2008) and 13 studies in the Atlantic and the Mediterranean Seas (Nickell and Moore, 1991; Kaiser and Spencer, 1996; Kenchington and Lundy, 1996; Moore and Howarth, 1996; Ramsay *et al.*, 1997; Veale *et al.*, 2000; Groenewold and Fonds, 2000; Bergmann *et al.*, 2002; Bozzano and Sardá, 2002; Link and Almeida, 2002; Jenkins *et al.*, 2004; Catchpole *et al.*, 2006; Martinez *et al.*, 2011) were reviewed.

The NE Atlantic studies were summarized in Table 4.1. Most studies were conducted in the North Sea, the Irish Sea or the Clyde Sea. When considering the top five scavenging species based on the numbers attracted to sampling gear by discards, six taxa occurred in >2 studies: *Buccinum undatum* (8), *Pagurus bernhardus* (8), *Asterias rubens* (6), *Cancer pagurus* (3), *Liocarcinus sp.* (4), *Carcinus maenas* (3). The small number of *Cancer pagurus* is likely due to the sampling methods, as is the underrepresentation of the fish guilds (See methods section below). Several factors influenced the number of identified key scavengers. Background densities and their spatio-temporal variation are likely the most influential, but were not always registered.

Seasonal and diurnal feeding patterns may affect food partitioning. Ramsay *et al.*, (1997) for instance illustrated that *Liocarcinus sp.* increase their scavenging activity during the night, while Nickell and Moore (1991) highlight that the monthly catch of *Pandalina brevirostris* and *Ophiocomina nigra* in the baited traps was correlated with variation in current speed over the spring/neap tidal cycle. Seasonal variation was not detected in Nickell and Moore (1991), but Groenewold and Fonds (2000) showed that the consumption rates varied due to temperature differences/seasons. Spatial variation in the segregation of food between species results from differences in scavenger assemblages by habitat type, and the resulting differences in competitive interactions (Ramsay *et al.*, 1997). The spatio-temporal variability of species' distributions is something which needs to be accounted for when developing indicators for species scavenging on discards.

Table 4.1 Discard scavenging studies in the NE Atlantic reviewed by WGECO.

	Bergmann <i>et al.</i> , (2002)	Catchpole <i>et al.</i> , (2006)	Groenewold and Fonds (2000)	Jenkins <i>et al.</i> , (2004)	Kaiser and Spencer (1996)	Martinez <i>et al.</i> , (2011)	Moore and Howarth (1996)	Nickell and Moore (1991)	Ramsay <i>et al.</i> , (1997)	Veale <i>et al.</i> , (2000)
Location	Clyde Sea	North Sea	North Sea	Irish Sea	Irish Sea	North Sea	Clyde Sea	Clyde Sea	Irish Sea	Irish Sea
Attraction time (h)	<48	2	<48	<96	?	<7	24	24	<76	<96
Baited traps	<i>Asterias rubens,</i> <i>Buccinum undatum</i> and <i>Neptunea antiqua,</i> <i>Carcinus maenas,</i> <i>Nephrops norvegicus,</i> <i>Pagurus bernhardus</i>	<i>Asterias rubens,</i> <i>Buccinum undatum,</i> <i>Liocarcinus depurator,</i> <i>Myxine glutinosa,</i> <i>Pagurus bernhardus</i>	<i>Asterias rubens,</i> <i>Buccinum undatum,</i> <i>Crangon</i>	-	<i>Asterias rubens,</i> <i>Buccinum undatum,</i> <i>Liocarcinus depurator,</i> <i>Myxine glutinosa,</i> <i>Pagurus bernhardus</i>	-	<i>Asterias rubens,</i> <i>Buccinum undatum,</i> <i>Carcinus maenas,</i> <i>Limanda limanda,</i> <i>Merlangius merlangus,</i> <i>Pagurus bernhardus</i>	<i>Buccinum undatum,</i> <i>Pagurus bernhardus,</i> <i>Natatolana puber,</i> <i>Pandalus borealis,</i> <i>Scopelochirus montagui,</i> <i>Scopelochirus hopei</i>	<i>Buccinum undatum, Colus gracilis,</i> <i>Hemimysis lamornae,</i> <i>Natatolana borealis,</i> <i>Pandalus borealis, Processa nouveliholthuisi</i>	-
Baited cameras or divers	<i>Brachyura,</i> <i>Cancer pagurus,</i> <i>Carcinus maenas,</i> <i>Crangonidae,</i> <i>Gobiidae</i>	-	-	<i>Callionymus lyra, Cancer pagurus,</i> <i>Flatfish, Majoidea,</i> <i>Ophiocomina nigra</i>	<i>Limanda limanda,</i> <i>Merlangius merlangus,</i> <i>Melanogrammus aeglefinus,</i> <i>Myxine glutinosa,</i> <i>Pleuronectes platessa</i>	-	-	<i>Buccinum undatum,</i> <i>Cancer pagurus,</i> <i>Liocarcinus spp.,</i> <i>Ophiura spp.,</i> <i>Pagurus bernhardus</i>	<i>Asterias rubens,</i> <i>Astropecten irregularis,</i> <i>Callyonimus lyra,</i> <i>Liocarcinus spp., Pagurus</i>	<i>Astropecten irregularis,</i> <i>Callyonimus lyra,</i> <i>Liocarcinus spp., Pagurus spp</i>

Top five scavenging species based on the numbers attracted by sampling gear

4.2.3 Sampling methods

Several methods have been employed to study the consumption of discards on the seabed. In the Northeast Atlantic baited traps have been used by a number of studies (Nickell and Moore, 1991; Moore and Howarth, 1996; Ramsay *et al.*, 1997; Groenewold and Fonds, 2000; Bergmann *et al.*, 2002; Castro *et al.*, 2005; Catchpole *et al.*, 2006). Baited cameras have also been used in a couple studies (Ramsay *et al.*, 1997; Jenkins *et al.*, 2004; Martinez *et al.*, 2011), and, to a lesser extent, divers (Bergmann *et al.*, 2002).

Several authors used a combination of *Nephrops* creels and funnel traps (Catchpole *et al.*, 2006; Nickell and Moore, 1991; Bergmann *et al.*, 2002). Nickell and Moore (1991) mention that the use of baited traps or creels to catch commercially important scavengers is an established method of sampling smaller species (e.g. amphipods) for research (Forbes and Hanley, 1853; Holdsworth, 1874 in Edwards, 1979; Paul, 1973; Shulenberger and Barnard, 1976 ; Ingram and Hessler, 1983). Groenewold and Fonds (2000) initially tested 10 different types of traps, and concluded that transparent tube traps, Danish prawn traps, and small (transparent plastic) amphipod traps appeared to be most suitable (details in Groenewold, 1999; Lindeboom and de Groot, 1998).

The pros and cons of the different types of creels and traps which have been used are listed in Table 4.2. The mesh size used in the traps is the main factor in determining the abundance and diversity of species retained. Small-meshed funnel traps or amphipod traps usually retain the smaller scavengers (amphipods and isopods) while the larger meshed traps retain larger organisms (molluscs, crustaceans and in some cases, fish). This subdivision of species scavenging on discards may cause bias in their observed abundances and therefore it will be difficult to determine the relative importance of different scavenging size groups as consumers of discards.

Table 4.2 Different types of traps used to sample benthic species scavenging in fisheries discards.

Trap type	Reference	Pros	Cons
Nephrops creel	Catchpole <i>et al.</i> , 2006; Nickell and Moore, 1991; Moore and Howarth, 1996; Bergmann <i>et al.</i> , 2002	Good for larger scavengers	Poor for smaller organisms (isopods and amphipods)
Crayfish trap	Catchpole <i>et al.</i> , 2006		
Funnel trap	Catchpole <i>et al.</i> , 2006; Bergmann <i>et al.</i> , 2002; Nickell and Moore, 1991	Good for smaller organisms (isopods and amphipods)	Poor for larger scavengers
Transparent tube traps	Groenewold and Fonds, 2000		
Danish prawn traps	Groenewold and Fonds, 2000		
Small (transparent plastic) amphipod traps	Groenewold and Fonds, 2000	Good for smaller organisms (isopods and amphipods)	Poor for larger scavengers
1m long 19-cm diameter pipe (6 cm flexible entrance)	Ramsay <i>et al.</i> , 1997	Good for larger scavengers (hermit crabs and whelks)	Larger animals may eat smaller scavengers while in the traps
1m long 19-cm diameter pipe (1.5 cm diameter flexible entrance)	Ramsay <i>et al.</i> , 1997	Excludes large predators	Poor for larger scavengers
90*40*45 cm (10 mm mesh size)	Castro <i>et al.</i> , 2005	Good for larger scavengers	
90*40*45 cm (1 mm mesh size)	Castro <i>et al.</i> , 2005	Good for smaller scavengers	

Due to the small size of entrances often used in traps (25–70 mm) the abundance of larger and more mobile scavengers such as fish are most likely underestimated (Groenewold and Fonds 2000; Catchpole *et al.*, 2006). Time-lapse camera observations can yield useful insights into on arrival times and residence time at food falls of these larger more mobile species (Kaiser and Hiddink, 2007) but they may not be optimal in obtaining accurate scavenger abundance of more mobile species. This is because time-lapse cameras only capture snippets of the feeding behaviour, thereby potentially missing the moments more mobile species are present. To obtain more accurate information on the scavenging behaviour of demersal fish Catchpole *et al.*, (2006) examined the stomachs of demersal fish for evidence of discarded material (Wieczorek *et al.*, 1999; ICES, 2000). With the introduction of cheaper underwater cameras the possibility to monitor the scavenging behaviour of demersal fish has increased.

4.3 Develop indicators of scavengers

Under a landing obligation the amount of organic matter redirected to the seabed is expected to be reduced. Furthermore, there is an additional source of carrion generated in the path of the trawl through direct physical disturbance which may also change following a landing obligation. A reduction in fishing effort, and consequently benthic impact, may occur in fisheries which are closed due to quota limitations on “choke” species. A choke species is defined as a species with the lowest quota in the mixed fishery which ‘choke’ the opportunity to catch the quotas of other species (Baudron and Fernandes, 2014). This may in turn affect species which have been observed to move into the path of a trawl to scavenge on dead or dying benthos. Alter-

natively, an improved selectivity may allow greater trawling activity. A substantial amount of carrion can be created through direct physical disturbance from trawling which varies with gear type and the habitat in which it is employed (Kaiser *et al.*, 2006). Understanding both the discard “subsidy” and the trawl path damage “subsidy” is important if we are to understand the ecological implications of fisheries generated carrion for the scavenging biota of benthic communities (Fonds and Groenewold, 2000; Kaiser and Hiddink, 2007).

The indicators developed below aim to be sensitive to changes in manageable pressures, for example, shifting from a landings quota system where discards are allowed to a catch quota system where discarding is prohibited. The indicators also aim to highlight where key scavenger species may experience a reduction in carrion available due to the changes in management. The indicators are expected to accommodate the fact that the sources of carrion (discards and direct physical damage from trawling) may be localized, creating hot spots of carrion.

Changes in the proportion of benthic biomass which are key scavengers may be important because the abundance of some scavenger and predator species (crustaceans, gastropods and sea stars) has been observed to increase due to fishery-induced changes (Rumohr and Kujawski, 2000). Rumohr and Kujawski (2000) observed a marked increase in records of starfish and swimming crab *Lioecarcinus holsatus*, and the frequency of occurrence of the common whelk. Greenstreet and Hall (1996) also reported that the abundance of scavenging flatfish such as dabs had also increased during the last century. Although such studies are useful in detecting long-term trends in scavengers, these changes are likely also linked to fisheries-induced reductions in competing species and top predators such as cod, as well as long-term changes in the environment (Kaiser and Hiddink, 2007).

Activity 1 – Investigate possible indicators for proportion of benthic biomass which are key scavengers

A term of reference will be constructed in collaboration with the Benthos Ecology Working Group (BEWG).

Activity 2 - Investigate possible indicators for biomass or abundance of key scavengers

A term of reference will be constructed in collaboration with BEWG.

Activity 3 – Investigate possible indicators based on important areas of spatial overlap of key scavenger species distribution and discards from main fleets

Together with the Benthos Ecology Working Group (BEWG) and the Study Group on VMS data (SGVMS), the abundance and spatial distribution of the key scavenger species, the general discard level and the spatial distribution of discards from the main fleets will be established, respectively. The spatial overlap of these distributions will be examined to highlight areas where key scavenger species may observe a food shortage under a landing obligation. This may lead to the development of a pressure indicator reflecting the reduction of introduced organic matter as a result of the landing obligation.

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5 Evaluate the ecological consequences of restoring stocks to MSY levels and the degree to which fisheries are “balanced” (ToR c)

The work on this multiyear ToR is planned as a meta-analysis of data from a series of ecosystems. These will be used first to determine the ecological consequences of restoring stocks to MSY levels. In addition we will evaluate the degree to which these fisheries can be considered as “balanced” (Garcia *et al.*, 2012), and the likely impact of fishing on these ecosystems. WGECO will consider changes to the ecosystem which are inadvertently manifested through management actions. E.g. there may be differences between the initial introduction of fishing as a generalist top predator and the current decrease in the effect of fishing to allow the rebuilding of natural top predators. Further, the way commercial exploitation attempts to constrain populations with specific bounds may change the development of cyclic patterns.

Some data are already available to the group and more will be collated for next year. The data should contain the information listed in Table 5.1.

Table 5.1. Data required for the meta-analysis of the ecological consequences of restoring stocks to MSY levels and the effect of the degree to which fisheries are “balanced” on the impact of fishing on the ecosystems.

Level	Type of data
Ecosystem	Name, proportion of approximate total biomass assessed (fish only), list of assessed stocks
Assessed stocks	Number-at-age by year, catch-at-age by year, fishing mortality-at-age by year, weight-at-age by year, and length distribution-at-age by year or if this is not possible, mean length-at-age by year (can be from a length weight relationship if not available otherwise)

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

6.1 Introduction

The Marine Strategy Framework Directive (MSFD) provides the legal imperative to promote ecosystem-based management across seas under the jurisdiction of the European Union (EC, 2008). Under the MSFD, Member States (MSs) of the European Union are required to achieve good environmental status (GES) for eleven Descriptors of GES by 2020. The EC 'Decision' document, intended to support MSs in their efforts to implement the MSFD, suggests the use of indicators at three levels, species, habitat and ecosystem (community), for Descriptor 1 "Biological diversity is maintained" (EC, 2010). At both species and habitat level, the 'Decision' document suggest the use of two types of distribution indicator; thus for criteria 1.1 "Species distribution" and for criteria 1.4 "Habitat distribution" MSs are supposed to consider indicators of "Distributional range" (1.1.1 and 1.4.1) and "Distributional pattern within the latter", where appropriate (1.1.2 and 1.4.2).

This focus on the distributional aspects of species' ecology is not restricted to just the European arena. The Canadian Species At Risk Act (SARA) requires assessment and protection of critical habitat for species at risk, similar to the Endangered Species Act (ESA) in the United States. The Federal Fisheries Act provides Fisheries and Oceans Canada with authority to protect fish and fish habitat essential to sustaining commercial, recreational and Aboriginal fisheries. Federal fisheries in the USA are subject to the Magnuson Stevens Fishery Conservation Management Act which requires that all fisheries management plans identify Essential Fish Habitat (EFH) and minimize the adverse effects of fishing on EFH. Federal regulations define EFH as "those waters and substrata necessary to fish for spawning, breeding, feeding, or growth to maturity". Policy directives that require measures for protecting habitats all imply an understanding of fish distributions in relation to environmental and habitat factors. (ICES 2013).

In this section the challenges facing distribution indicators are reviewed and the attributes of distributions which should ideally be monitored and their value to management are discussed. The definition of aspects was approached through analysis of a relatively rare species, cuckoo ray, in the continental shelf waters of the Celtic Seas and Bay of Biscay MSFD regions. By visually exploring the distribution of Cuckoo ray, attributes of the distribution that are of scientific and management interest were determined and indicators that accurately reflect changes in these attributes were suggested. Issues linked to sampling effectiveness are addressed and effects on indicator performance and interpretation of having a large land mass, Ireland, situated within the species distribution is approached by focusing our study on the Celtic Seas subregion.

6.2 Challenges facing the development of distribution indicators

Many univariate metrics have been developed and used to monitor change in both the total area a species occupies (termed its range in the MSFD, EC, 2010) and variation in its abundance at different locations within this space (termed its distributional pattern within the range in the MSFD). In this section some of the challenges that

have to be addressed in order to identify operational distribution indicators that can be used within an assessment, advice and management context are addressed.

6.2.1 The abundance – distribution relationship

The abundance-occupancy relationship is one of the most generally held patterns in ecology (Gaston *et al.*, 2000; Blackburn *et al.*, 2006), both within species (as a species increases in abundance, so its range increases) and between species (more abundant species occupy larger ranges than rare species). Both intra- and interspecific relationships have been demonstrated in fish populations (Overholtz, 2002; Hinz *et al.*, 2003; Fisher and Frank, 2004; Blanchard *et al.*, 2005; Frisk *et al.*, 2011). The relationship appears intuitive, as a population doubles in size it requires double the space to contain it, but in reality it is more complicated as abundance usually increases at a faster rate than the size of the occupied range (Gaston *et al.*, 2000). Density-dependent habitat selection, the expansion of a species into more marginal habitats as abundance increases and its contraction into optimal habitat as abundance declines (Fretwell and Lucas, 1970; Partridge, 1978; MacCall, 1990), has also been demonstrated in fish populations (Hinz *et al.*, 2003; Blanchard *et al.*, 2005). Within a species full potential range therefore, the fraction of the total area occupied at highest densities when population abundance is high are likely to be regions of prime habitat for the species in question, and consequently that part of the full range most likely to be occupied when population abundance declines to low levels (MacCall, A.D. 1990, Holt *et al.*, 2002, Holt *et al.*, 1997, Freckleton *et al.*, 2006; Frisk *et al.*, 2011).

This body of ecological theory predicts that metrics/indicators of fish population distributional range, distributional pattern within range, and abundance should all covary in a predictable way, implying a likely high level of redundancy among them several indicators providing information on a single signal (Greenstreet *et al.*, 2012a; Lyashevskaya and Farnsworth, 2012) when other factors remain constant. Since the MSFD also requires indicators of species abundance this has led some authors to question whether there is a need for distribution indicators as well (Greenstreet *et al.*, 2012b).

6.2.2 Geographical scale

The MSFD defines specific spatial scales, the region or the subregion, at which GES is to be assessed. In the Northeast Atlantic Ocean region, for example, four distinct subregions have been defined, (i) the Greater North Sea, including the Kattegat, and the English Channel; (ii) the Celtic Seas; (iii) the Bay of Biscay and the Iberian Coast; and (iv) in the Atlantic Ocean, the Macaronesian biogeographic region, being the waters surrounding the Azores, Madeira and the Canary Islands. The Mediterranean Sea is also divided into four separate subregions: (i) the Western Mediterranean Sea; (ii) the Adriatic Sea; (iii) the Ionian Sea and the Central Mediterranean Sea; and (iv) the Aegean-Levantine Sea. The Baltic Sea and the Black Sea constitute two further regions, neither of which have specified subregions.

Many fish species have distributional ranges that greatly exceed the area of individual regions and subregions (Figure 6.1). The two distribution related indicators specified for Criterion 1.1. “Species distribution” in the ‘Decision’ document, “Distributional range” (1.1.1) and “Distributional pattern within the latter, where appropriate” (1.1.2), are supposed to be species level indicators. Reporting changes in these two characteristics of species distribution at just the MSFD subregion scale, par-

ticularly in the Northeast Atlantic, is unlikely to reflect actual changes taking place at geographic scales relevant to the species concerned.

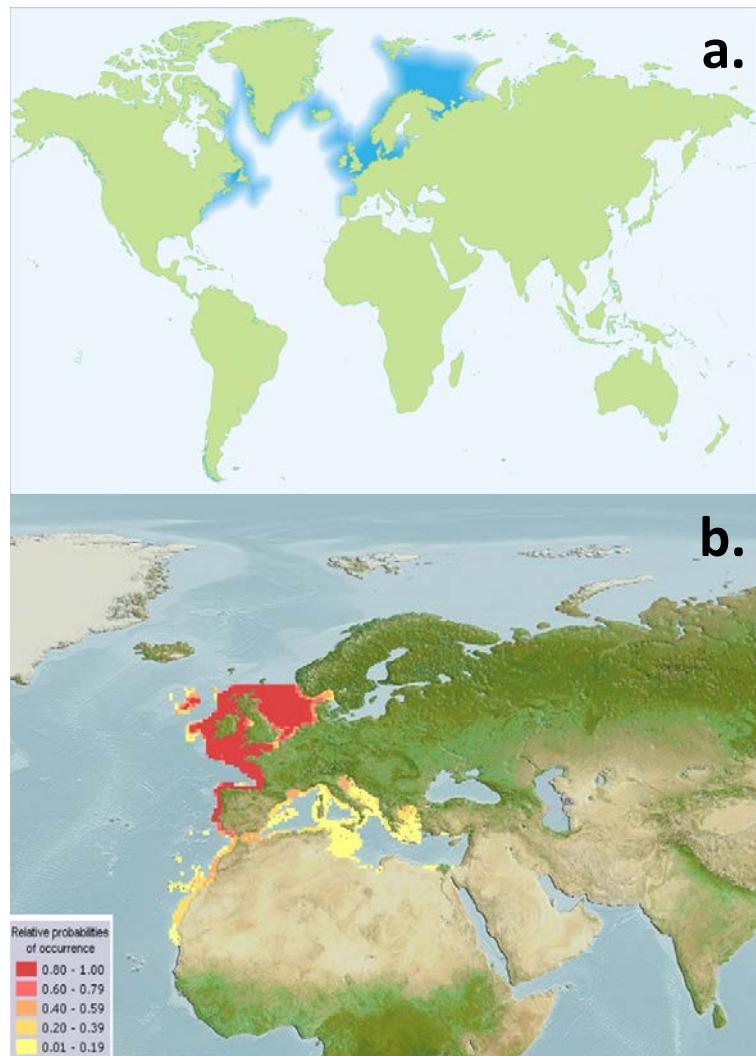


Figure 6.1. The global distributions of a) cod *Gadus morhua*, which spans four MSFD subregions of the Northeast Atlantic and extends across the Atlantic Ocean to the Canadian, Greenlandic, Icelandic and North American seabards and b) cuckoo ray *Leucoraja naevus*, which spans at least three MSFD subregions of the Northeast Atlantic and also extends into the Mediterranean Sea (http://upload.wikimedia.org/wikipedia/commons/6/60/Gadus_morhua-Atlantic_cod.png; <http://www.fishbase.org/Summary/SpeciesSummary.php?ID=4326andAT=cuckoo+ray>).

6.2.3 Distribution as a surveillance indicator for population abundance

Distribution shifts are the most obvious response by fish populations to changes in sea temperature regime (Brander *et al.*, 2003), with warming trends tending to cause northward shifts in range (Rose, 2005). Recent warming of the Northeast Atlantic has coincided with northerly shifts in the distribution of many species generally associated with more southerly latitudes (Quéro *et al.*, 1998; Perry *et al.*, 2005; Beare *et al.*, 2004). However, these shifts in distribution are often only really apparent as changes in distribution when the entire distributional range of the species is mapped. If the assessment focus has a smaller geographic scale than the distributional range scale of the species in question, then such distributional shifts actually manifest themselves as changes in population abundance within each contiguous assessment areas.

Herring, near the south of their range in the Celtic Sea, declined in abundance during warming periods but increased during cooling. Conversely, sardine, at the north of their range, displayed opposite abundance trends (Southward, 1980; Alheit and Hagen, 1997). This switching appears to have been occurring since the 13th century (Southward *et al.*, 1988). During a warm phase of the Atlantic Multidecadal Oscillation herring stocks in the English Channel, where this boreal species is close to the southern edge of its range, declined markedly (Southward, *et al.*, 1988; Edwards *et al.*, 2013), while at the same time in the Nordic seas, where herring are close to the northern edge of their range, their biomass increased by a factor of 10 (Toresen and Ostvedt, 2000). When the AMO reverted back to a cooler phase in the 1970s, the herring population in the Norwegian Sea declined by four orders of magnitude, but the latest AMO warming phase has again seen an increase in herring biomass back to 1960s levels (Edwards *et al.*, 2013).

6.2.4 Types and ease of interpretation of metrics used in the literature

Spatial configuration refers to the spatial properties and arrangement, position, or orientation of habitat patches within the broader survey area. Spatial ecology has always been an important dimension for the interpretation of ecological phenomena and there are many indices for representing spatial configuration (e.g. Fortin and Dale, 2005; Maguire *et al.*, 2005). The identification of significant coral and sponge habitats in Eastern Canada conducted by Kenchington *et al.*, (2010) were based on a geospatial model (Kenchington *et al.*, 2014) using kernel density analysis to identify significant concentrations of corals and sponges. Metrics describing coral and sponge habitat were later constructed from individual patches (e.g. mean patch area and shape) using attributes of their statistical distribution (e.g. mean, maximum, variance) of the corresponding patch variable (e.g. size, shape) (Kenchington *et al.*, 2012). The spatial relationship among patches, or patch configuration can also be quantified using nearest-neighbour and other statistics capturing information on the relative position of the patches within the survey landscape (fragmentation). When evaluating trends in these indicators it is important to select a constant area for the analyses and to make comparisons with surveys using a single trawl gear (Kenchington *et al.*, 2012). This will reduce the effect of catchability on the analysis.

Several types of metrics have been suggested including, for example:

Patch area and density: The area occupied by a species can be expressed using summary statistics drawn from the population of habitats or patches in the broader survey landscape (e.g. mean, median, maximum, variance, etc.). Patch density is the number of patches per unit area. Patch perimeter is usually highly correlated with Patch area but in some cases may have better distributional properties and so could be seen as an alternative to Patch area. These statistics can be compiled within a lat/long/depth grid system and monitored over time.

Nearest neighbour measurements: Isolation/Proximity: Isolation or proximity refers to the tendency for patches to be relatively isolated in space from other patches. If d_{ij} is the nearest-neighbour distance from patch i to another patch j of the same type, then the mean nearest-neighbour distance over all patches is a measure of relative isolation. Connectivity: Isolation/Proximity of patches can be interpreted through connectivity. If ecological or oceanographic knowledge governing the dispersal of gametes or larvae is available then the information can be used to predict a neighbourhood size that reflects a gamete dispersal range or other ecological process. The

number of patches that fall into the neighbourhood size could then become a measure of connectivity.

Dispersion: Dispersion refers to the tendency for patches to be regularly or contiguously distributed (i.e. clumped) with respect to each other. Dispersion can be calculated for patches, tow locations with specific bycatch or individuals.

The range of potential metrics available to fulfil the two MSFD distribution indicator roles is extensive and in many cases, quite what these metrics convey regarding actual changes in the distribution of an organism is far from transparent. A number of metrics have been used in the recent literature to capture variation in distributional range; these include (e.g. Rindorf and Lewy, 2012; Greenstreet *et al.*, 2012b):

The proportion of samples containing more than a ‘fixed’ number of individuals. Where this ‘fixed’ number is zero this is effectively the number of samples that give a “presence” score.

The proportion of empty samples.

The proportion of structurally empty samples.

Interest also focuses on the geographical spread of individuals within the range using metrics such as:

The average distance to centre of gravity.

The area of the contour ellipse.

Other metrics have been used to describe differences in the level of aggregation of individuals across the distributional range; these include (e.g. Rindorf and Lewy, 2012; Greenstreet *et al.*, 2012b):

Lloyds index of patchiness.

Sample abundance mean/variance ratio

Measures based on points on the Lorenz curve, such as height of the curve at points that account for a specified proportion of the population.

Measures based on the area below the Lorenz curve i.e. the Gini and the Atkinson measures.

Described thus, the majority if these metrics require scientific expertise in their use and interpretation. To ensure that distribution indicators used in the MSFD are more readily comprehensible to stakeholders, it would be desirable if the metrics used to convey change in distributional range and pattern within the range were more easily understood.

Some metrics are calculated using raw sample data, while others are based on the parameters fitted to the data using maximum likelihood function which may require that statistical distributions of the population are known in advance.

Many of these indicators would fail a criterion for good indicator performance linked to capacity to communicate what change in the indicator actually means. In many cases, managers and scientists alike are primarily concerned about identifying core areas of a species distribution and knowing about the fraction of the population that these areas contain. In most instances, these distribution core areas will not be contiguous so that information on the number of units, their size distribution, and the distance between units will often be wanted to provide an overall evaluation of ‘fragmentation’.

6.2.5 Sampling effects for species encountered infrequently in surveys

Species over which there is the greatest conservation concern are frequently among the rarest encountered in surveys. Often majority of samples in any survey will contain no individuals of the species in question while just a few samples may contain 1 to a very few individuals. Such data are extremely difficult to model using geostatistical techniques. Spatial semi-variograms used to support parameterise geostatistical methods for example might indicate no significant spatial structure in the data. Under such circumstances, the most appropriate ‘surface’ to fit to the data is a flat one equal to the mean of all the values. Often such statistical approaches are not applied to such datasets and instead distributional metrics are applied to the raw sample data. But this implicitly assumes a spline fit to the data, such that the fitted density at each sample location equals the observed sample density. Such an approach implicitly assumes a spatial structure in the data where a geostatistical approach would infer that no such structure existed. It may be necessary to investigate distribution data for several years together in order to increase the number of non-zero observations.

6.3 Is there a need for distribution indicators?

WGECO discussed need for distribution indicators in detail based on annual distribution maps of cuckoo ray (see section 6.4). Examining these maps and drawing on examples from North Atlantic and Baltic waters covering fish and benthic organisms, the following conclusions were made:

- Comparison of a time-series of annual distribution maps is a challenging exercise and to be tangible, it must be reduced to distilling key elements of the distribution, ideally captured in the form of a single univariate metric, which can then be plotted as a time-series graph. The problem lies in identifying these key elements and, having done this, devising effective metrics that capture the information of interest and can convey it in an intelligible manner that is can support analysis.
- The geographical scale on which the distribution is examined should correspond to the distribution range of the population. Examining distribution of two distinct populations using a common metric does not provide useful results.
- There can be changes in geographical distribution which are not caused by abundance. For example, range may decrease as a result of changes in hydrographic conditions in specific areas and even a large increase in abundance may not lead to a repopulation of these areas. Conversely, if new areas become inhabitable, abundance may potentially increase beyond current levels. In these cases, carrying capacity of the range may have changed and hence there may be a need to revise reference levels based on this.
- Even in the event where range remains constant, the population may exhibit a change in relative abundance in different parts of the range. This could be the case where the stock distribution pattern changes through fishing pressure, hydrographical change or changes in growth conditions in different areas. In such cases, the population may become more aggregated in space and hence more vulnerable to pressures in specific areas. Alternatively, a less aggregated population may disperse out of de facto refuges (section 9.7) and become more sensitive to the pressure exerted in other areas.

In conclusion, WGECO found that distribution indicators can be potentially be very important to management. The indicators should reflect aspects of both the distributional range (the area occupied by a species at any point in time) and the distributional pattern within the range (core areas of high abundance). Three properties of each aspect were considered relevant: the areal extent, the geographic location (e.g. north/south or shallow/deep changes in distribution) and fragmentation/isolation. In total, 6 aspects should thus be captured by indicators: extent of the occupied range, geographic placement of range, fragmentation/isolation of the occupied range, extent of the core area, geographic placement of the core area and fragmentation/isolation of the core area. Ideally, all indicators should reflect the distribution of the population and not be restricted to specific management areas.

6.4 Modelling Cuckoo ray distribution in the Celtic Seas and Bay of Biscay

6.4.1 Data collection

Distribution maps for the area were derived by combining four surveys (the French, Irish, Northern Irish, and Scottish bottom-trawl surveys). This produced a preliminary dataset sufficient to produce distribution maps to be used as a basis for the development and testing of candidate distributional indicators.

The Scottish, French and Irish bottom-trawl surveys all use a Grande Overture Verticale (GOV) trawl fitted with a 20 mm codend liner. The Northern Irish survey uses a rock-hopper gear, also with a 20 mm codend liner fitted. The French and Irish surveys follow stratified random designs over the period that we consider, while the Scottish survey switched from a rectangle stratified design to a stratified random design in 2011 and the Northern Irish survey uses a fixed station stratified design. All four surveys were carried out in the fourth quarter (Q4) of the year. The Irish Ground Fish Survey (IGFS) is carried out by the Irish Marine Institute on the Celtic Explorer. The French groundfish survey in the Celtic Sea and Bay of Biscay is carried out by Ifremer using Thalassa. The Scottish West Coast VIa groundfish survey is carried out by Marine Scotland using Scotia III. The Northern Irish survey is undertaken by AFBI using the Corystes. All surveys follow the protocol for the Standard International Bottom-trawl Surveys (IBTSWG, 2010) trawling at a speed of 4 knots for 30 minutes. All fish caught are identified and measured according to the IBTS protocol.

6.4.2 Data cleaning and calibration

Comparison of abundance at length data in areas where trawl surveys overlapped suggested similar catchability of cuckoo ray in all the surveys used in the current study (Trenkel, pers. Comm.). Survey catch data per unit time trawled were converted to estimates of density on the ground (numbers per km²) after calculating the area of seabed swept between the wings of the trawl. Where tow distance was not recorded this was estimated as the straight line between the shoot and haul positions, and where estimates of average distance between the wings of the trawl were not provided, these were estimated using linear regression models as functions of depth, door-spread, and warp length depending on which parameters were available (Greenstreet *et al.*, 2012).

6.4.3 Geostatistical analyses

The ArcGIS10.2© software package was used for mapping and spatial analyses. All trawl location latitude and longitudes were transformed into 'northing' and 'east-

ings' on a zone 29 universal transverse Mercator projection and their positions plotted to define the spatial extent of the surveyed area. The surveyed area was defined by 'drawing' a boundary around all trawl stations collected over the period 2008 to 2013. The total area thus defined extended between 'northing' 4860989 m to the south and 'northing' 6670989 m to the north and between 'easting' 1111947 m to the east and 'easting' 266947 m to the west, was 414 103 km² in extent with a boundary of 8164 km (Figure 6.2). The Spatial Analyst toolbox was used to derive a 5 km by 5 km grid so that each pixel for which cuckoo ray density estimates would be interpolated coved 25 km². Interpolated distribution surfaces were derived using the Geostatistical Analyst toolbox, using a Radial Basis Function which is an exact interpolator as it forces the surface to pass through each measured sample value. Unlike other similar methods RBF allows predictions to be higher and lower than the measured values. This method is appropriate to large datasets where the smoothed surface is not expected to show marked irregularities. - All cuckoo ray size classes were mapped as a single group. A six year period was selected on the basis that these surveys provided the most consistent spatial coverage. For the purposes of indicator development and testing, this period was considered sufficient and the inclusion of additional years of data with more limited spatial coverage would probably have distorted our analysis.

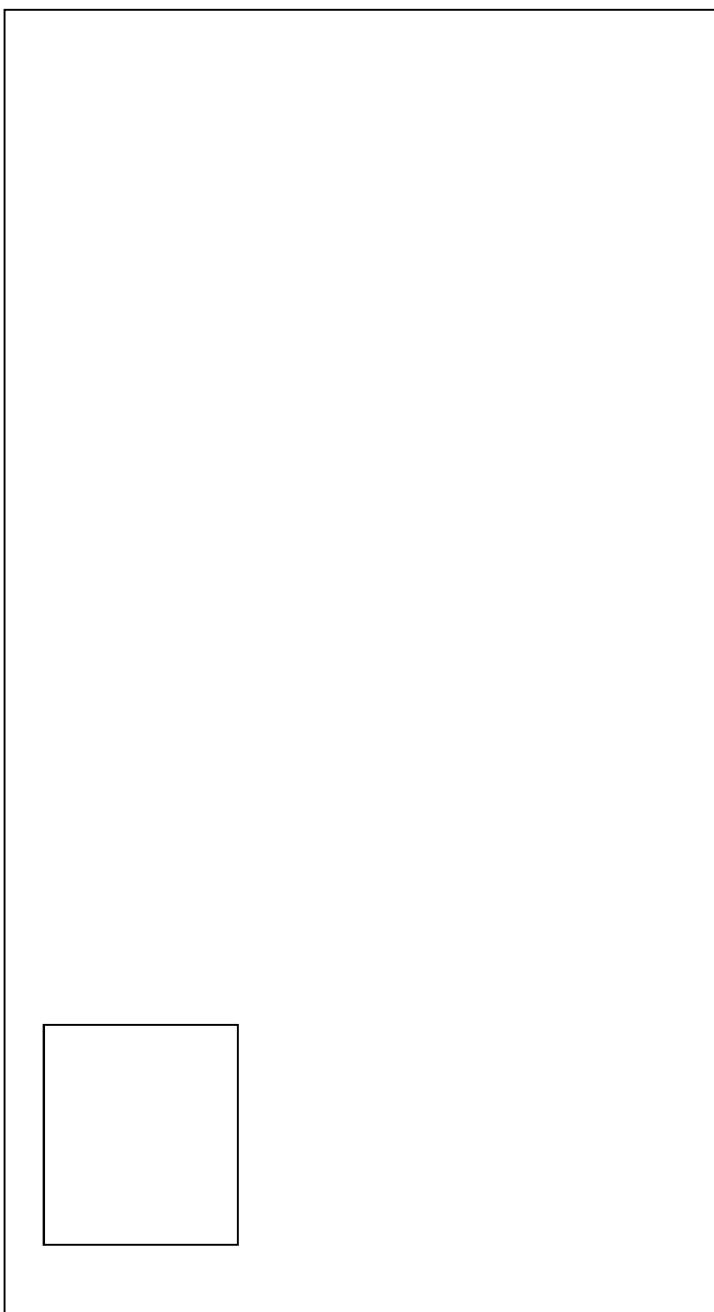


Figure 6.2: IBTS trawl location sampled between 2008 and 2013, showing spatial coverage of the 4 cruises. Overlap occurs with the Irish and Scottish survey in the north and the Irish - French in the south. The pale yellow area which covers an area of $11\,650 \text{ km}^2$ (~3%), is referred to as the 'Irish Sea' in the analyses and the pale orange covering an area of $29\,250 \text{ km}^2$ (~7.6%), is called the 'West'. The 'North' is the grey area from Scotland to the north of Ireland with an area of $125\,475 \text{ km}^2$ (~32.6%) and the grey area to expanding as far as the Bay of Biscay, the largest area covering $218\,000 \text{ km}^2$ (~56%), is call 'South' in the analyses. The entire surveyed area is $383\,375 \text{ km}^2$.

A Thin Plate Spline kernel function was used; neighbourhood searching was standard, to include at least 3. Thin plate spline (TPS) is a deterministic radial basis interpolation technique with a local stochastic component. Whaba (1990) describes the theory behind thin plate splines, the TPS fit resists bending, this implies a penalty involving the smoothness of the fitted surface.

TPS allows a straightforward analysis of spatial data. There is no need for prior estimation of the spatial dependence of measurement points. The thin plate spline has a number of properties which contribute to its reputation: It produces smooth surfaces which are infinitely differentiable, there are no free parameters that need manual tuning, it has closed form solutions for warping and parameter estimation and there is a physical explanation for its energy function.

A characteristic of the interpolation process when so many of the actual sample density estimates were zero is to generate a large number of both negative and low value positive pixel density estimates. While negative values are clearly an artefact of the interpolation processes, many of the low-value positive pixel values were also interpolation anomalies due to the smoothing of grid node estimates between spatially distant positive and zero value sample densities. Actual sample density estimates were posted on top of the gridded surface and the interpolated grid values examined manually. In some instances it was clear that the interpolation process had generated some low value false positives ($>1 \text{ km}^{-2}$ and $< 5 \text{ km}^{-2}$) at grid node locations where the actual sample data provided no evidence that these should be anything other than zero. In such cases, the pixel values concerned were edited and given a zero km^{-2} density value. By the nature of the interpolation process, the interpolated surface had negative density values at some grid nodes. To counter this problem, all grid node density estimate values $< 2 \text{ km}^{-2}$ were arbitrarily assigned a value of zero. A value of 2 km^{-2} was chosen so as to generate sharper demarcation between occupied and empty grid nodes. Setting a constant limit of this type ensures that occupied area is tightly linked to overall abundance (Swain and Sinclair, 1994).

6.4.4 Distribution maps and abundance

Figure 6.3 shows the interpolated distribution of cuckoo rays, as density (number km^{-2}) in each year of the survey. Knowing that each pixel represented 25 km^2 of sea area, multiplying these densities by 25 provided estimates of the abundance of cuckoo ray in each 25 km^2 pixel. Summing across all pixels provided an estimate of total cuckoo ray abundance across the whole surveyed area (Figure 6.4). This abundance trend revealed marked year-to-year variation which is expected in trawl a survey. A decline from ca. 3.6 million individuals in 2009 to ca. 2.3 million individuals the following year represents a mortality loss of 33% in one year. The return to a population abundance of ca. 3.6 million individuals one year later therefore represents a high level of change which could be due to a number of sources (e.g. recruitment, migration, sampling error). In 2010 approximately half the population exceeded the length-at-maturity (Maia *et al.*, 2012) and assuming a 50–50 sex ratio this means that there was approximately 575 000 mature females in the population. If the observed change was due to recruitment, the recruitment of approximately 1.3 million juveniles the following year represents a contribution of approximately 2.3 recruits per mature female. Given that an average female might produce approximately 40 to 45 eggs (Maia *et al.*, 2012), this level of recruitment is feasible but probably represents higher than average recruit survival. Figure 5 shows the length frequency distributions of fish sampled in the four surveys. These data show little evidence of the high levels of recruitment that would be necessary to generate the abundance increase observed in 2011.

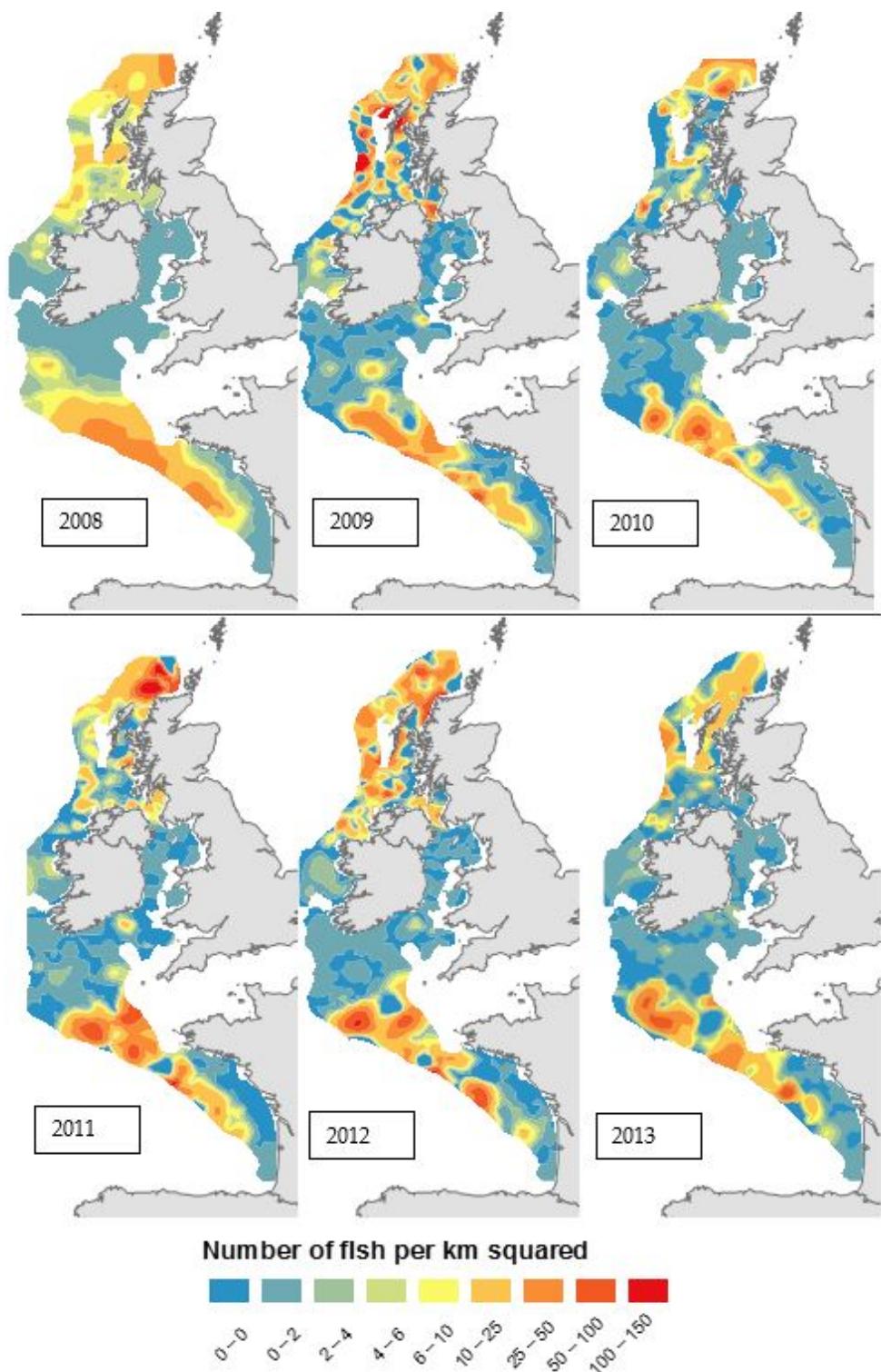


Figure 6.3. Spatial variation in the abundance if the cuckoo ray within the defined survey area in each year 2008-2013.

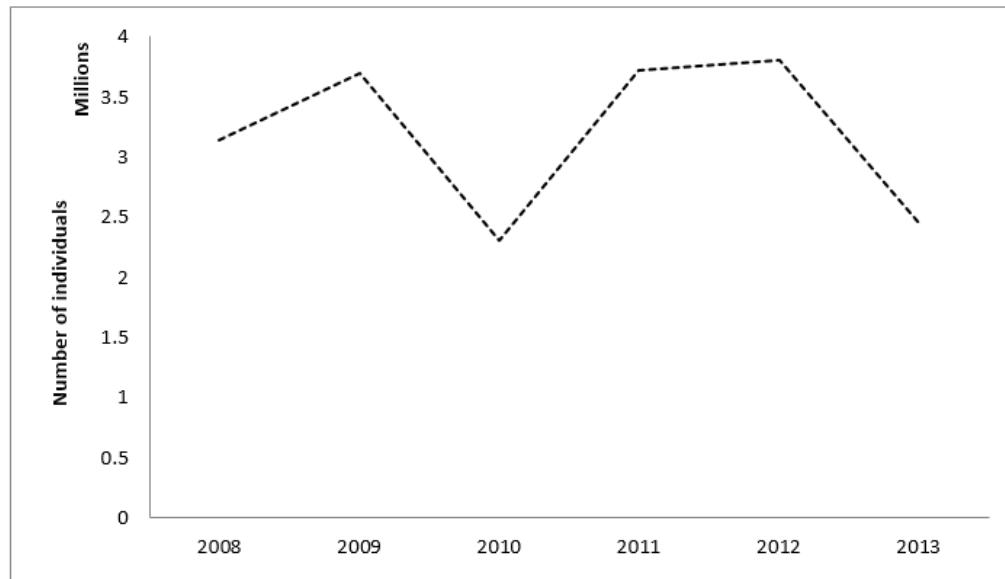


Figure 6.4. Time-series of the total abundance of the cuckoo ray across the entire survey area over the period 2008 to 2013.

6.5 Identifying key elements of distribution and selecting appropriate metrics

We interpret distributional range to mean the area occupied by a species at any point in time. Thus assuming that distribution range and population abundance are related, a species' distributional range might be expected to vary as population size varies. Furthermore, the area occupied by a species, its distributional range, might alter even with no change in population size. This change in range location might simply be linked to random 'wobble', or it may reflect a geographical shift in range location. Thus the maximum potential range of a species might actually exceed the area occupied by the species at any one time, even when population size is high. However, metrics of range geolocation are now necessary in order to differentiate between random 'wobble' in a species distribution range, or a systematic shift in its location; an indicator of range geolocation.

The metrics investigated are listed in Table 6.1. These capture most of the characteristics of distribution that are discussed above, although other metrics of dispersion could be further investigated.

Table 6.1. Metrics investigated.

INDICATOR	DEFINITION	METRIC USED
Distributional range	The actual area occupied by a species at any point in time.	'total area occupied'
Maximum potential range	All area that has ever been occupied by a species across entire survey period.	'cumulative total area occupied'
Range geolocation	Maximum, minimum and median 'northing' and 'easting' and depth of the Distributional range.	Variation in the max, min and median distributional range
Distribution core area	The area within the distributional range that holds the highest densities of individuals and which account for a specified proportion of the population (e.g the smallest fraction of the distributional range that holds 50% of the population).	'smallest area occupied by Dx'
Core area – number of units	The number of contiguous patches or units of distributional core area	'smallest number of patches required to contain Dx'
Core area – unit size frequency distribution	The size frequency distribution of core area patches	'min, mean and max size of core area patches'
Core area – connectivity median inter-unit distance	The median nearest-neighbour distance between distributional core area units	'min, median and max distance between core areas'

6.5.1 Distributional Range

We use the 'total area occupied' by cuckoo ray in each year as the metric to monitor interannual change in the distributional range of cuckoo ray. The 'total area occupied' is defined as the actual number of 5 km² by 5 km² pixels occupied by cuckoo ray at a density >2 km⁻². This is visually represented in the distribution maps shown in Figure 3. Using the 'total area occupied', the variation over the period 2008 to 2013 can be quantified and is shown in Figure 6.5. The 'total area occupied' dropped sharply between 2008 and 2009 and then remained relatively constant thereafter. We also plot the 'cumulative total area occupied'. In essence this additional metric provides information on the area that has ever been occupied by cuckoo ray as each successive annual distribution map is taken into consideration. The 'cumulative total area occupied' shows a gradually reducing increase towards an asymptotic value regardless of actual variation in 'total area occupied' between years. For example, although the 'total area occupied' decreased sharply between 2008 and 2009, the 'cumulative total area occupied' still increased; new areas were being occupied in 2009 that in 2008 were unoccupied.

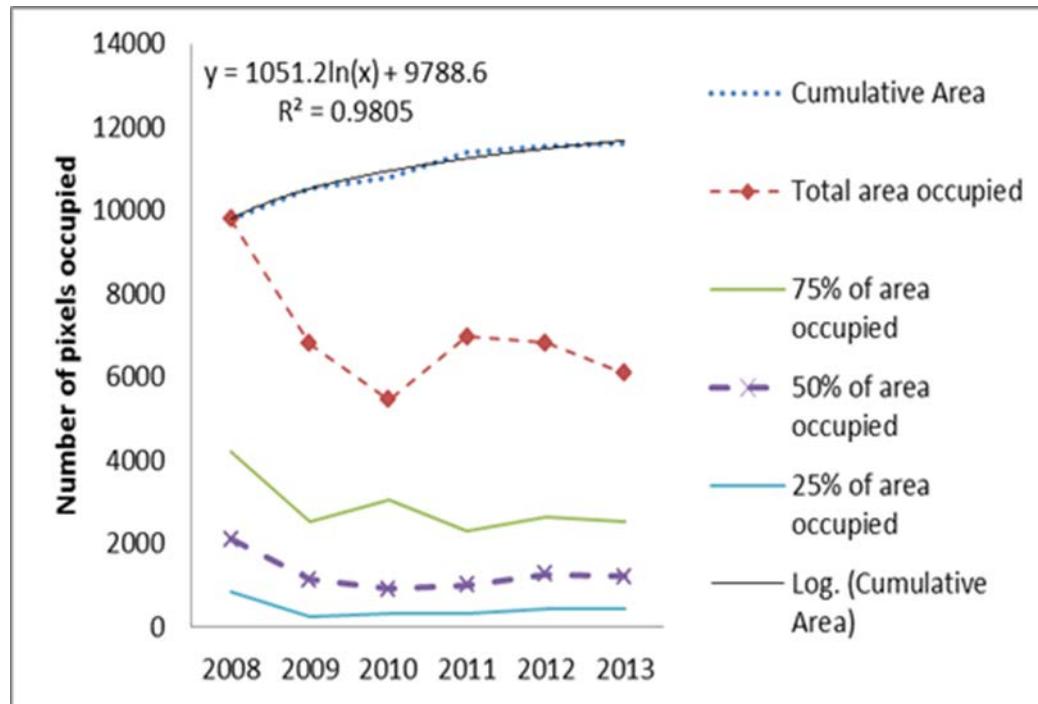


Figure 6.5: Temporal variation in distributional range and three pattern within the range metrics over the period 2008 to 2013. The ‘total area occupied’ is the metric used to convey change in distributional range. We also show the ‘cumulative total area occupied’, aggregation of the ‘total area occupied’ over time (cumulative area). This metric shows the total area ever occupied by cuckoo ray as each successive annual distribution map is included in the analysis. The Gleason semi-log model is shown fitted to the ‘cumulative total area occupied’ trend (Log. (Cumulative Area)). For distribution pattern within the range we show the smallest number of pixels that includes 75%, 50% and 25% of the population in each year.

Similar to species – area curves, modelling the increase in ‘cumulative total area occupied’ as successive annual distribution maps become available could provide managers and policy groups with information regarding the theoretical maximum distribution range (under prevailing environmental conditions) that the species in question might ever occupy. For example, Arrhenius power and Gleason semi-log functions, both of which have been used to explain species – area curves (Greenstreet and Piet, 2008), to our six years of cuckoo ray distribution data provided near identical fits (Arrhenius: $r^2=0.9813$; Gleason: $r^2=0.9805$). Our total surveyed area extended over 16 567 pixels and both models suggest that, had 25 years of data been available, the total area ever occupied by cuckoo ray would be between 13 172 pixels (Gleason model) and 13 453 (Arrhenius model), approximately 80% and 81% respectively of the total surveyed area. The cumulative total area occupied after six years of data was actually 11 597 pixels; approximately 87% of these theoretical maxima. Neither function, however, reaches an asymptote and assuming that at least some of the area within the total surveyed area constitutes habitat that is unsuitable to cuckoo ray, fitting an asymptotic function, such as a Michaelis-Menton function, might be more appropriate (Figure 6.6).

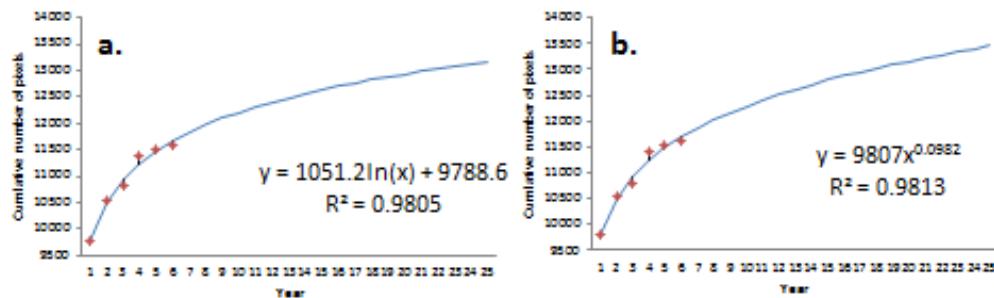


Figure 6.6. Fitted Gleason (a.) and Arrhenius (b.) cumulative total area occupied models.

To derive metrics of the distributional pattern within the range, we first determined three preliminary parameters; these are the number of smallest number of pixels that accounts for 25%, 50% and 75% of the population in each year, i.e. the most densely populated pixels in the distribution. In each year, occupied pixels were ranked (by density, highest density given rank 1) and the number of highest density pixels counted until the required fraction of the population had been accounted for. Variation in these basic parameter values is shown in Figure 6.5. Dividing the number of pixels accounting for each fraction of the population by the 'total area occupied' in each year provided an estimate of the proportion of the distributional range holding each specified fraction of the population. A one-to-one relationship between these two proportion values, i.e. 25% of the population in 25% of the area, 50% of the population in 50% of the area, 75% of the population in 75% of the area, implies a perfectly evenly distribution of the species in question. Increasingly downward shift in observed values in plots with proportion of the population on the x -axis and proportion of the range on the y -axis indicates increasing levels of aggregation in the species' distribution. Figure 6.7 shows such a plot for cuckoo ray in our survey area for the period 2008 to 2013. The cuckoo ray distribution in each year was clearly aggregated and the degree of aggregation varied from year to year. The fraction of the 'total occupied area' that accounted for 50% of the population varied between 15% in 2011 and 22% in 2008.

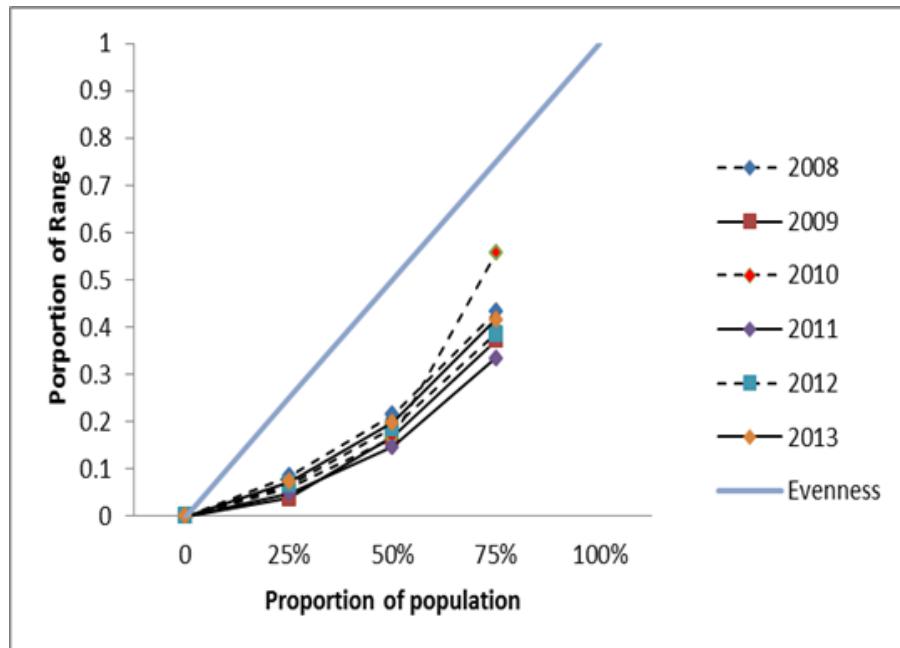


Figure 6.7. Variation in the three aggregation metrics in each year. The null hypothesis relationship associated with and even distribution is shown for comparison.

Using just these three simple aggregation metrics as indicators of distributional pattern within the range fails to convey critical information that is clearly apparent from the distribution maps; namely that the cuckoo ray population in this region consists of two quite spatially distinct subunits. Figure 6.8 consolidates this impression. This figure was constructed by first determining the proportion of the population located at each pixel, thereby standardizing the data against variation in total abundance (Figure 6.4), then taking the average of these proportions across all six years. The clear gap between the southern northern spatial subunits, spanning a distance of approximately 165 km, is much more emphatic when considering relative abundance rather than absolute abundance.

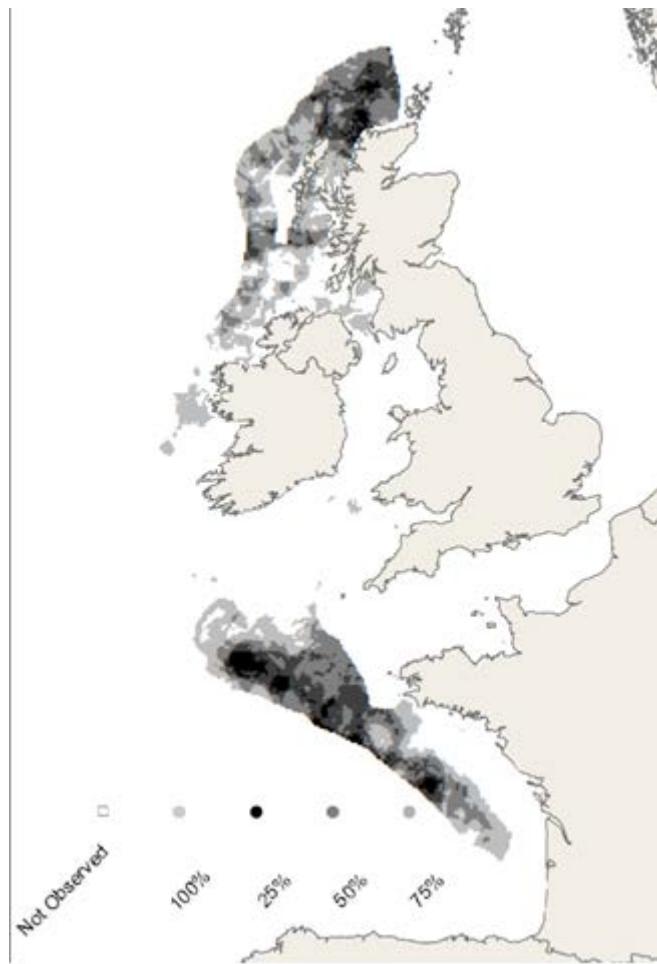


Figure 6.8. Relative abundance of cuckoo ray averaged across each year over the period 2008 to 2013.

Clearly metrics of distributional pattern within the range applied to the whole population would convey an entirely misleading impression, such a center of gravity distribution ellipse defined by 1 standard deviation in both dimensions fitted to probability of encounter data (Figure 6.9). Having established that distinct spatial subunits exist, the obvious next step is to examine each subunit separately, repeating all the analyses undertaken so far. Trends in cuckoo ray abundance within each spatial subunit were similar; the most obvious difference being apparent in 2008 when abundance in the northern subunit was relatively low compared with abundance in the southern subunit (Figure 6.10). Comparisons of the numbers of cuckoo ray sampled at each length class in each subunit and year suggested that in all years, the southern subunit contained a greater fraction of smaller sized fish in the 25 cm to 40 cm size range (Figure 6.11). Trends in temporal variation in the ‘total area occupied’ varied between the two subunits, but the ‘cumulative total area occupied curves’ were similar. The number of pixels accommodating 25%, 50% and 75% of the population were similar in both subunits, particularly in 2010 and 2012 when population abundance in the two subunits was almost identical (Figure 6.12a). On average the level of aggregation in both subunits was similar, but interannual aggregation was greater in the northern subunit (Figure 6.12b). This was further emphasized when actual aggregation metrics (A) were derived following $A = \frac{R-P}{P}$ where P is any specified proportion of the population and R is the proportion of the ‘total area occupied’

holding the specified proportion of the population (Figure 6.12c). Essentially this equation estimates the extent of deviation below the even distribution line. The metrics varies between 0 (completely even distribution) and approach 1 (highly aggregated distribution). Figure 6.13 summarizes this information. The level of aggregation was more variable in the northern subunit, but aggregation was generally greater in the southern subunit.

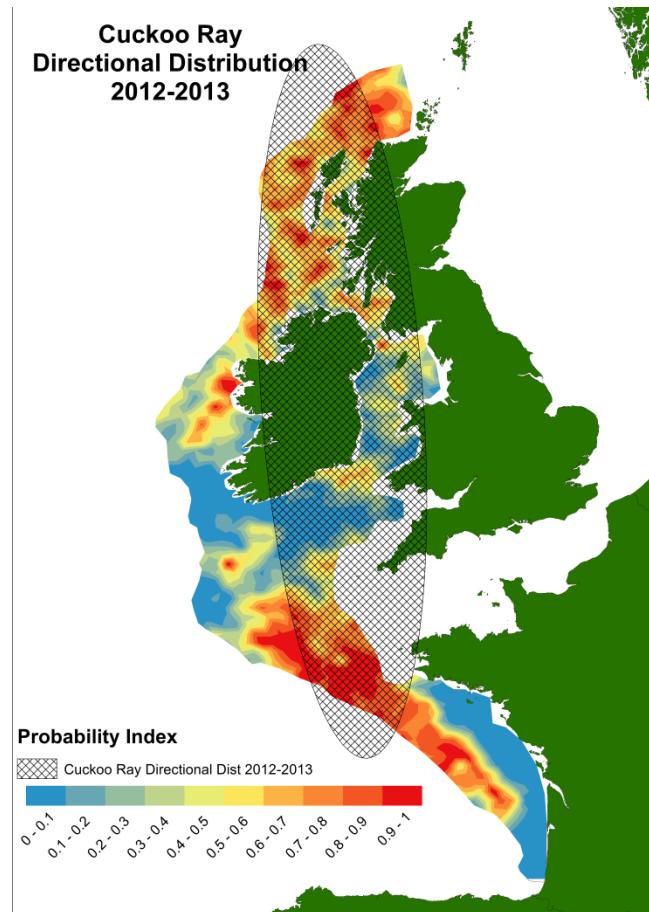


Figure 6.9. Centre of gravity ellipse determined as \pm one standard deviation either side of the mean centre of gravity in each dimension fitted cuckoo ray probability of encounter data.

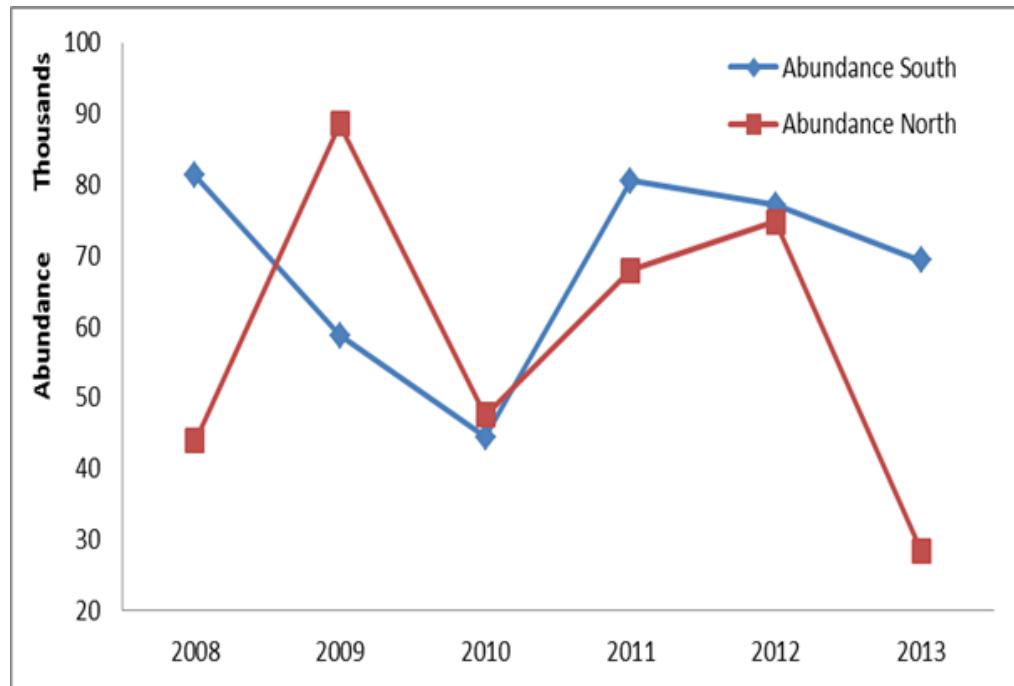


Figure 6.10. Temporal trends in the total abundance of cuckoo ray in the southern and northern spatial subunits.

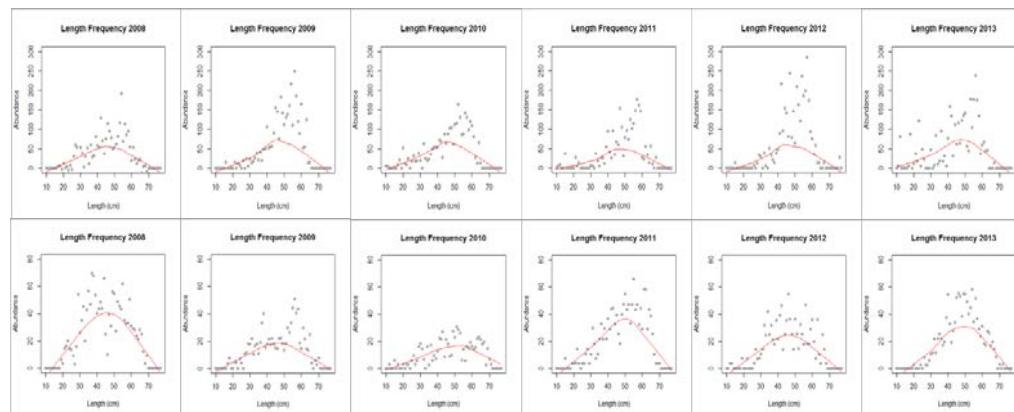


Figure 6.11. Length frequency distributions of sampled cuckoo rays from each spatial subunit in each year. Red lines show loess smoothers fitted to the data. Northern subunit is along the top and southern subunit along the bottom.

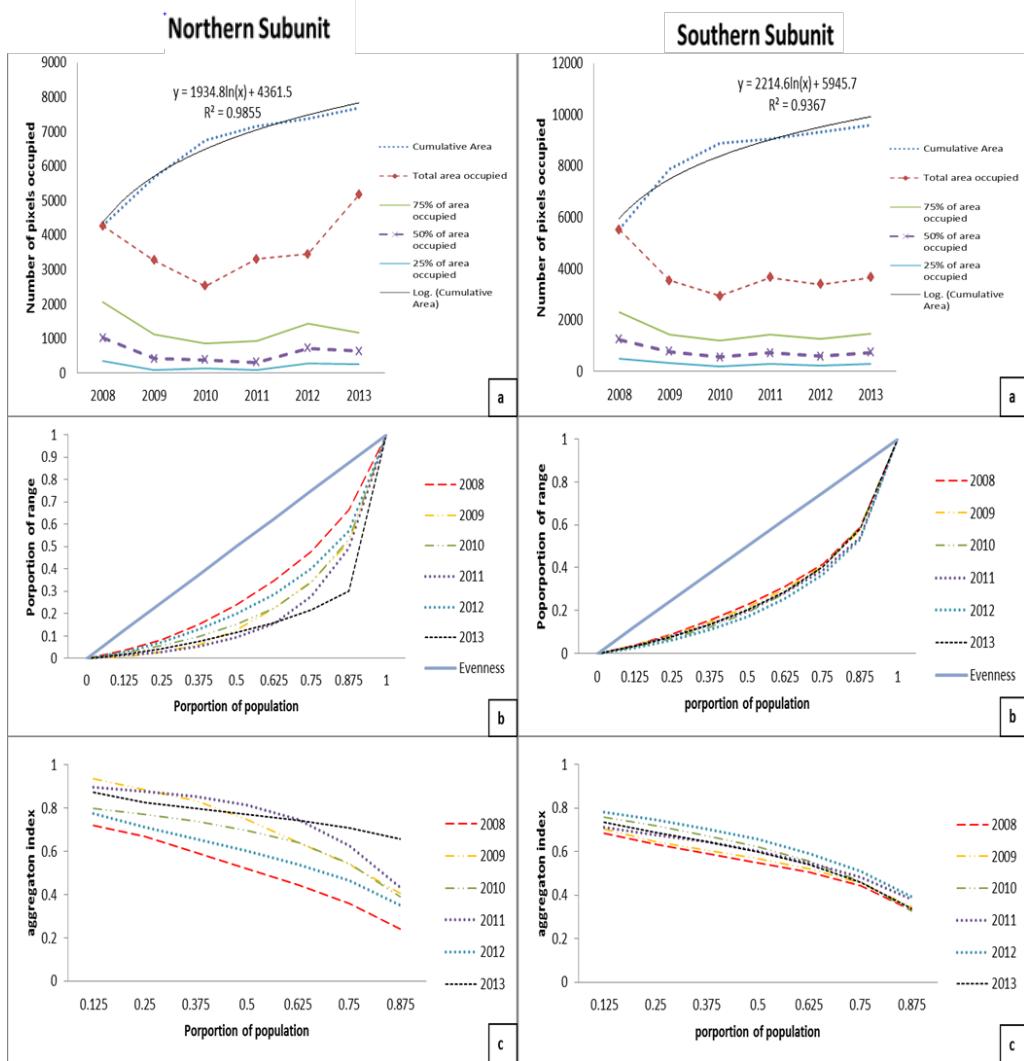


Figure 6.12. For both the northern and the southern spatial subunits: a) shows temporal variation in the, 'total area occupied', 'cumulative total area occupied', and the three basic distribution aggregation parameters (see Figure 6.5 and text for more details); b) shows temporal variation in three aggregation metrics (see Figure 6.7 and text for more details); and c) temporal variation three aggregation indicators used to describe distributional pattern within the range (see text for details).

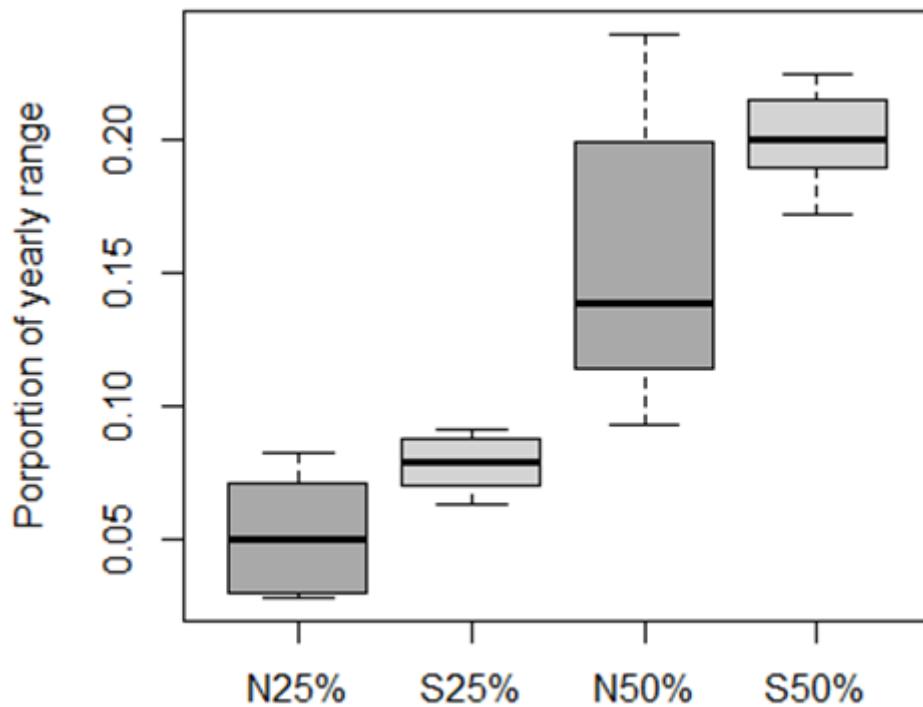


Figure 6.13. The proportion of the ‘total area occupied’ by 25% and 50% in the northern (dark grey) and southern (light grey) cuckoo ray spatial subunits. Heavy horizontal lines indicate the median values averaged across all six years, boxes indicate the positive and negative data quartiles and whiskers indicate the data range.

We now define “core area” within the cuckoo ray distributional range as the smallest number of pixels containing 25% of the population. This was converted to an estimate of the actual area by multiplying the number of pixels by the area (25 km^2) of each pixel. Over the period 2008 to 2011, consistent trends were apparent over the entire population and in each of the subunits (Figure 6.14a). The marked decline in the extent of “core area” observed in all three trend lines from 2008 to 2009 did not coincide with any major decline in abundance. Between 2011 and 2012, the trend line for the northern subunit differed from the southern (Figure 6.14a). Fragmentation of the “core area” was more variable in northern subunit and this variability was reflected in the level of fragmentation across the entire population. This variability in fragmentation of “core area” in the northern subunit seemed unrelated to changes in the actual extent of “core area”. In the southern subunit, as the extent of “core area” increased and decreased, so did the number of individual fragments.

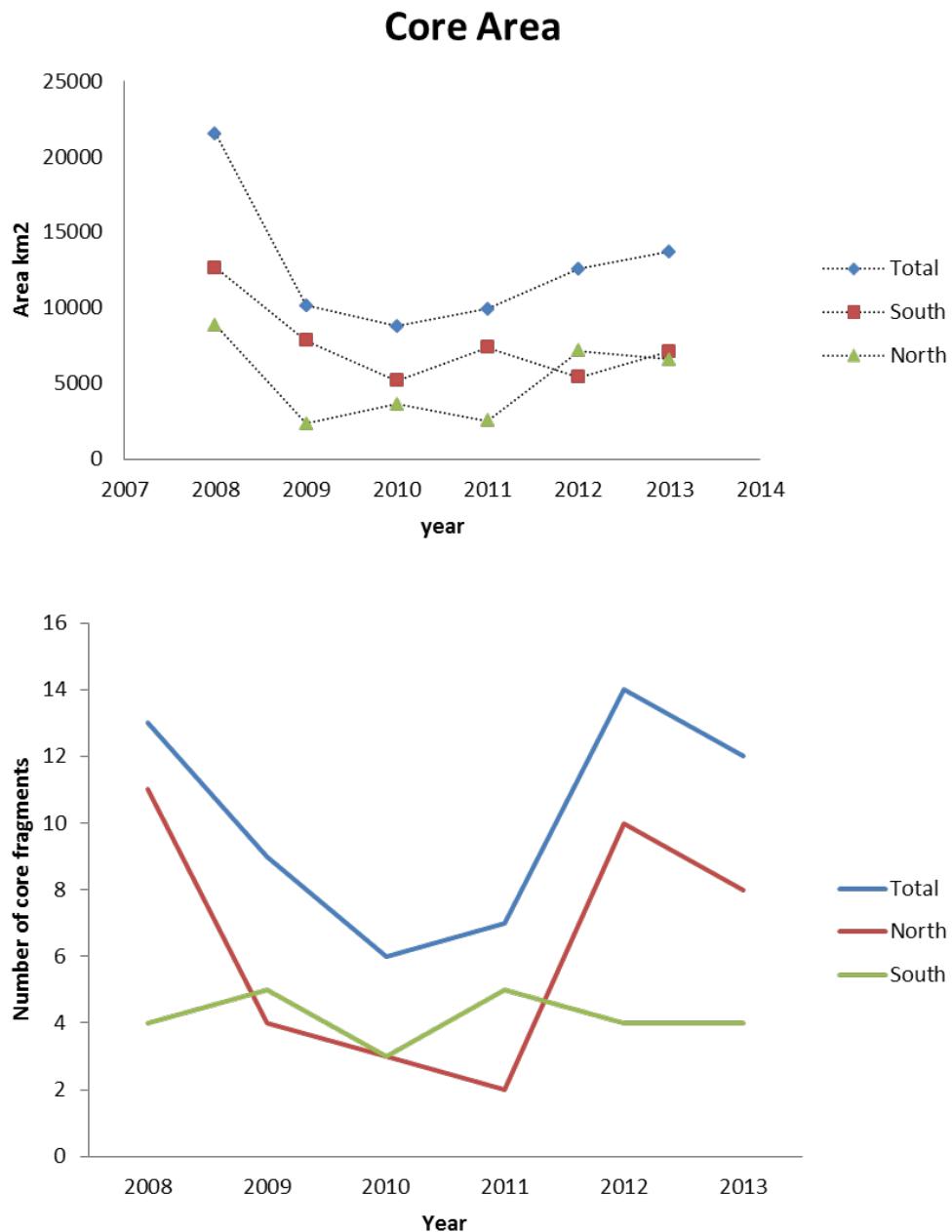


Figure 6.14. Temporal trends in: (a) the actual area (i.e. number of pixels multiplied by 25 km²) holding 25% of the cuckoo ray population across the whole population and within each spatial subunit; and (b) the number of individual fragments of contiguous adjacent pixels that make up the “core area” of the distribution.

We next looked for any systematic shift in the distribution ranges of both the northern and southern subunits. Density data at each pixel was converted to presence/absence so that any shift in the location of occupied pixels was examined. This looked for any change in the geographic position of the distributional range with no change in the distributional pattern within the range. No obvious systematic trends were apparent, but in the northern subunit there was some indication of interannual variation in the median pixel location.

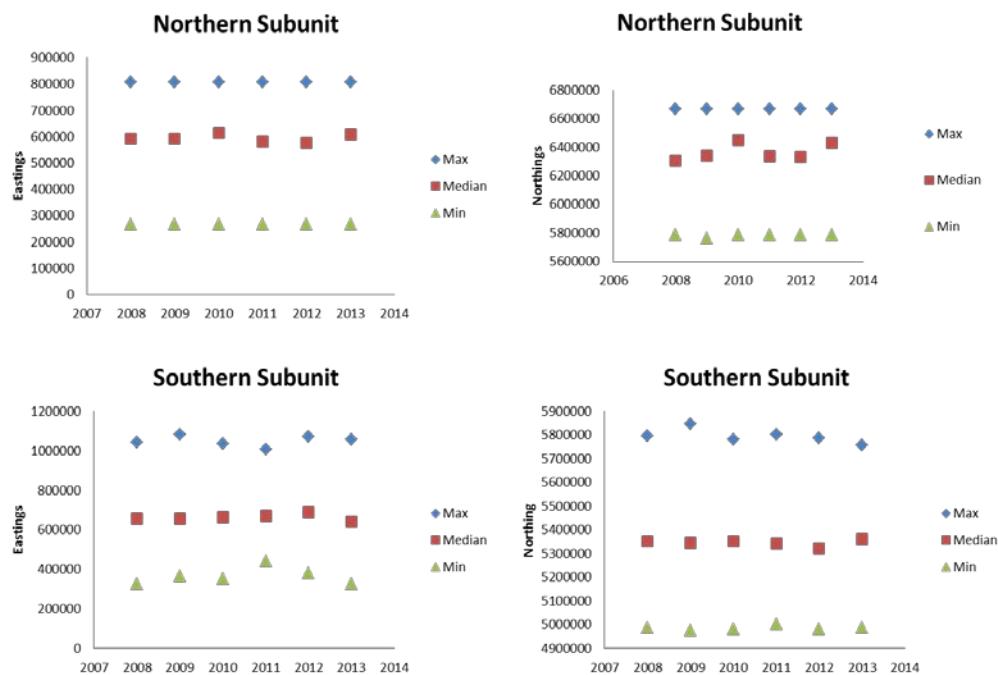


Figure 6.15. Interannual variation in the median maximum and minimum ‘northings’ and ‘eastings’ for the northern and southern cuckoo ray subunits.

6.5.2 Conclusions from analyses of Distributional Range

Understanding how the distributions of fish species change over time is a key aspect of delivering an operational ecosystem approach management. Our preliminary data exploration concludes that using spatially interpolated maps is a necessary first step in capturing patterns in distribution. This allows spatially distinct subunits to be addressed separately. Following this approach revealed different trends between the two cuckoo ray subunits in ‘total area occupied’, the metric used to track the extent of the distributional range. It also suggested that aggregation and measure of the distributional pattern within range also varied between the two subunits. Aggregation was consistent in the south but more variable in the north. This emphasizes the need to examine maps of the species distribution so that the approach to analysing species distribution patterns can be varied on a species-specific basis. The dispersion parameter described here is essentially similar to the Lorenz curves, however the dispersion parameter is conceptually simpler to convey to stakeholders and policy-makers, making it a more appropriate metric to use in supporting MSFD implementation. Further analyses will allow us to examine what the dispersion parameter tells us about the carrying capacity. By examining the juveniles and adults separately we may discover that some of the variation we have observed may be explained by distributional differences between different age/size classes of cuckoo ray.

These metrics have yet to be tested to ensure they are not biased. Using these indicators without such testing could lead to the conclusion that a change in distribution has occurred when in fact there has only been a change in bias.

It is perhaps unlikely that a systematic geographic shift in the distributional range of a Lusitanian species like cuckoo ray will have occurred since the surveyed area may not capture the southern distribution of the species. The metric may be of more value when looking at species in which the northern and southern range is captured.

Major drivers of change in the distributions of fish include anthropogenic pressures such as fishing (Walline *et al.*, 2012, Garrison and Link, 2000) and changing environmental conditions, which could be linked to climate change (Perry *et al.*, 2005). A next step could be to look at how the cuckoo rays “core areas” correspond to the habitat, in depth, salinity and substrate. By comparing the cuckoo ray to other ray species, we may discover areas that are unsuitable for all species and areas that are unsuitable to some species due to competition. Linking species distribution to fishing effort would be valuable in determining risk to species.

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

7.1 Distribution

Distribution indicators are derived from point or transect sample data. In this report we have focused on describing fish distribution using groundfish survey trawl sample data. It is an obvious truism that the more samples that are available, the greater the confidence in the resulting distribution maps and indicators. But beyond this, consistency in sampling becomes an important issue. In this report WGECO has based distribution maps on the data derived from four surveys and issues regarding differences sampling efficiency and methodology have had to be addressed. Deriving reliable distribution maps and indicators that are relevant at geographic scales that match species' distributional range would be greatly facilitated if survey methodology, gears, timing, etc., was rationalized to greater degree. Variation between years in the area covered by surveys also poses a problem. One either has to limit the extent of the area analysed to the minimum area covered by all surveys or rely on extrapolation to "fill in" the edges in years when survey spatial coverage is reduced. Such extrapolation will inevitably introduce bias in the resulting distribution maps and indicators. In an ideal world survey spatial coverage should be sufficient to contain the entire range of the species in question in order to avoid edge effects. Persistent survey spatial coverage and consistency in methodology will all serve to reduce the risk of interpolation artefacts unduly influencing results.

7.2 Surveillance

The concept of using surveillance indicators to support the use of operational indicators may have data requirement implications if new monitoring programmes, or additional analyses of data provided by current monitoring programmes. As a rule though, metrics currently being considered as surveillance indicators are supported by ongoing monitoring programmes and they are often already routinely calculated, but the metric has just not achieved the status of an operational indicator. Using these metrics as surveillance indicators should therefore primarily make use of a resources that is already available.

7.3 Sensitive species

Determining which fish species are sensitive relies heavily on knowing their life-history traits. Currently, for many species these traits have to be estimated following a defined procedure, but access to actual species-specific and area-specific data on life-history traits would be beneficial. This approach has primarily been developed in the North Sea. Access to the necessary data in other marine area would be strongly advantageous in rolling that approach across other marine regions and subregions covered by the MSFD. Expanding this approach out to cover marine taxa also has obvious data requirement implications.

8 Surveillance indicators and their use in implementation of the MSFD.

8.1 Introduction

The European Union (EU) Marine Strategy Framework Directive (MSFD) is based on the concept of Good Environmental Status (GES), which involves protecting the marine environment, preventing further deterioration and restoring it where practical, while using marine resources sustainably (EC, 2008). Ecosystem state is monitored and assessed using ecosystem indicators and associated reference levels and targets. Indicators used to support implementation of the MSFD were initially expected to be ‘operational’; i.e. to have well understood relationships between state and specified anthropogenic pressure(s) and to have defined GES targets. Operational indicators track change in the state of particular attributes of specific ecosystem components, and recovery to GES should be achievable by the introduction of specific measure(s) to manage the known pressure(s). More recent discussion has highlighted the potential value of ‘surveillance’ indicators (Morris and Cole 2002; Whitfield and Elliott, 2002; Smart *et al.*, 2003; Cury and Christensen, 2005). WGECO defines these as indicators that monitor key aspects of the ecosystem for which there is either insufficient evidence to define targets, thus limiting their capacity to support formal assessment, or where the links to anthropogenic pressures are not well understood, so limiting their capacity to underpin specific management advice. Surveillance indicators are not expected to directly track state in relation to GES, but may provide tangential information that informs and supports science, policy and management.

The Activity-Pressure-State-Response (APSR) concept is well established as a framework for assessing causes, consequences and responses to change in the marine environment (e.g. Greenstreet *et al.*, 2009; Reiss *et al.*, 2010). The APSR framework identifies specific *pressures* that ensue from different human *activities*. Operational ecosystem indicators are used to track changes in *state* that might be caused by the changes in *pressure* to which they are subjected. Any change in *state* beyond defined reference points will trigger a management *response*. Selection of indicators should follow a clearly stated process, and indicators should be sensitive to pressure, responsive to management action and specific to the pressure of interest (Rice and Rochet, 2005; Piet *et al.*, 2008). Although surveillance indicators may not necessarily have clearly understood pressure-state relationships or defined management response levers, they could perform an important function within any APSR framework that might be applied to support MSFD implementation.

In this section a structure for adding a surveillance indicator evaluation process into the APSR management framework is developed. Examples of metrics that could perform surveillance indicator roles and support implementation of the MSFD and fisheries management are then discussed. Finally, criteria against which the performance of different potential surveillance indicators can be evaluated are considered, which can be applied to guide selection of the most effective surveillance indicators. We focus here on the APSR framework because, in explicitly including *activity*, this has distinct advantages over the pressure-state-response (PSR) (Garcia and Staples, 2000) indicator based framework (Greenstreet *et al.*, 2009; Reiss *et al.*, 2010). Other frameworks, such as the Driver-Pressure-State-Impact-Response framework, have also been widely used (Gimpel *et al.*, 2013; Knights *et al.*, 2013). Here the use of the term *driver* is frequently synonymous with meaning of *activity* in the APSR framework.

Our proposals for adapting the APSR framework can therefore easily be transposed to modify the DPSIR and PSR frameworks if necessary.

8.2 Expanding the APSR framework to accommodate surveillance indicators

In many cases, it should prove possible to manage important ecosystem components using operational indicators within APSR management frameworks. For example, fisheries management uses such a framework based on indicators of landings or effort (*activity*), fishing mortality (*pressure*), stock biomass (*state*) and total allowable catch (*response*) (Greenstreet *et al.*, 2009; Piet *et al.*, 2010; Probst *et al.*, 2013). Similarly, the large fish indicator (LFI) has been developed to monitor the broader impact of fishing on the state of the whole demersal fish community. For several marine regions, targets have been determined for the LFI and well defined relationship between variation in fishing mortality and changes in the LFI have been established (Greenstreet *et al.*, 2011; Shephard *et al.*, 2011; Modica *et al.*, 2014). Seabird breeding success has been used as an important state indicator both from a foodweb perspective, as an indicator of the availability of prey, and from a population dynamics perspective. Kittiwake (*Rissa tridactyla*) breeding success has been related to variation in winter sea surface temperature. Targets can therefore be set for chick production in each breeding season based on observed environmental variation. Furthermore, the deviation from expected breeding performance associated with industrial fishing has also been established (Frederiksen *et al.*, 2004). In other instances, well-developed indicators might well be available with which to monitor change in key components of marine ecosystems. However, either because they do not have defined targets or because their pressure – state relationships are ill-defined, so cannot support formal assessment or inform specific management measures, these indicators cannot be used within an APSR framework and so cannot be considered as operational indicators.

For example, The MSFD ‘Decision’ document suggests EU member states (MSs) develop indicators of trophic guild biomass to monitor change in the structure of marine food webs (EC 2010). Such indicators have been available for over a decade and used to monitor change in foodweb structure in both the Greater North Sea and Celtic Seas MSFD subregions (Greenstreet *et al.*, 1997; Heath, 2005a; Heath, 2005b). Recent workshops to consider revision of the ‘Decision’ document have increased emphasis on the need for trophic guild indicators (ICES 2014a; 2014b). Consequently, fish trophic guild biomass indicators have been developed specifically to support MSFD implementation (Greenstreet *et al.*, Submitted a). These are based on the ICES coordinated first quarter (Q1) international bottom-trawl survey (IBTS) data; the same dataset used to derive the North Sea LFI. Whereas targets for the LFI can be set on the basis that the size composition of North Sea demersal fish was consistent with sustainable exploitation at the start of the time-series in 1983, the same logic does not apply for foodweb structure. There is no evidence to suggest whether foodweb structure at this time was at GES. Many human activities probably exert pressure on marine foodwebs, but these diverse pressure – state relationships have not been established. These trophic guild indicators convey important information regarding changes taking place within marine foodwebs, but because the structure of marine foodwebs at GES in each MSFD subregion is not well known, targets cannot be set for these indicators. Furthermore, even if targets could be defined, it is not clear how human pressures on these foodwebs should be managed in order to achieve them. These foodweb trophic guild biomass indicators cannot therefore be used within an

APSR management framework and so cannot be considered to be operational indicators.

Foodwebs provide the fundamental structure linking the organisms that make up ecosystems. It is inconceivable that human activities in the marine environment can be managed to achieve GES across the whole marine ecosystem without taking cognizance of changes occurring in the foodweb. Variation in foodweb structure has a bearing on the management of other components of marine ecosystems. A major increase in piscivorous fish might represent increased predator competition for forage fish (Reilly *et al.*, 2014), and so affect targets set for seabirds and marine mammals and influence management of industrial fisheries. Likewise a major increase in benthivorous fish would suggest increased predation pressure on benthic invertebrates (Heath, 2005a), and could infer that targets set for seafloor integrity and benthic biodiversity might need revision. Even if foodweb trophic guild indicators cannot be used directly within an APSR management framework, they can clearly perform a surveillance role, influencing how APSR frameworks are used in respect of operational indicators applied to other ecosystem components.

Figure 8.1 shows how the use of operational indicators to support APSR management frameworks might be modified to incorporate the use of surveillance indicators. Provided the surveillance indicator remains within defined bounds, the APSR framework applied to the operational indicator in question operates as usual. However, should the surveillance indicator move outside these defined bounds, then this should trigger a policy reaction.

The methods which can be used to define bounds depend on the amount and type of information available (ICES WKMSFD4-II 2015). Where data and knowledge available is very limited, for example, when sampling has only just begun, appropriate limits can be suggested based on expert knowledge from similar ecosystems, theoretical considerations or a desired direction of change. Where data exists, no undesirable effects have been observed but knowledge of the direct relationship between the indicator and other ecosystem characteristics is limited, the indicator limits should describe the observed range of known indicator values. Where data exists and undesirable effects on other ecosystem components has been observed or is predicted based on solid knowledge of the direct relationship between the indicator and other ecosystem aspects, the range of indicator values associated with no substantial undesirable effects on other components could be used to set limits that denote the desirable range of indicator values. The presence of substantial undesirable effects on ecosystem components can be determined based on a variety of measures. Ideally, the evaluation includes both expert judgment, analyses of historic data and investigation of model results.

Figure 8.2 illustrates how the surveillance indicator bounds might be defined where data exists and no undesirable effects have been observed. In this example the bounds are set as the upper and lower limits of variation in the value of the surveillance indicator over the duration of the time-series. If this represents a foodweb structure indicator for example, it would suggest that no major changes in foodweb structure had occurred that had profound implications for other ecosystem components. Alternatively, movement outside these bounds represents change that goes beyond scientific experience and knowledge, and so comprises capacity to provide sound reliable scientific advice.

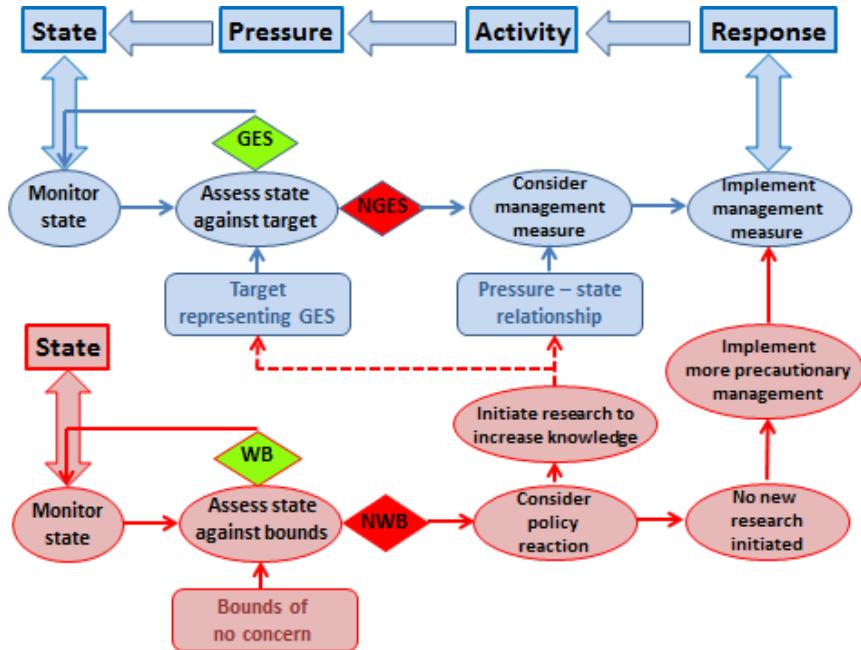


Figure 8.1. Diagram illustrating how surveillance indicators (red process) can complement operational indicators (blue process) in an Activity-Pressure-State-Response approach to the MSFD. Operational indicators evaluate whether state is meeting (GES) or failing (NGES) 'Good Environmental Status' targets. Surveillance indicators evaluate whether state is within bounds (WB) or not within bounds (NWB), where these bounds represent the upper and lower limits of a range in state for which there is no 'specific cause for concern' (see Figure 8.2).

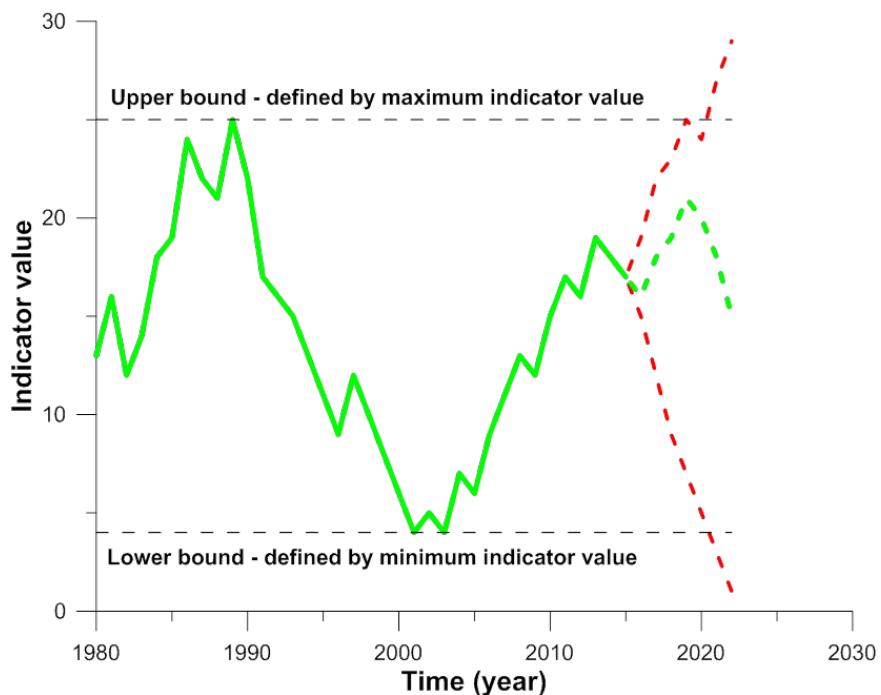


Figure 8.2. Schematic of a generic surveillance indicator time-series showing historical upper and lower bounds within which the indicator has varied over the time-series duration (dashed black lines). Variation in indicator values beyond these bounds (e.g. dashed red trajectories) infers that the ecosystem component in question is changing towards a state not previously experienced. Such a situation would represent a 'specific cause for concern' and should trigger a policy reaction since, under these circumstances, the knowledge necessary to underpin reliable scientific advice would increasingly be in short supply (see Figure 8.1).

At this point knowledge needs to be expanded, so the policy reaction could be to initiate targeted research programmes. These would have to address two key issues in respect of each APSR management framework likely to be affected:

1. Do targets set for operational state indicators need to be re-evaluated so that the outcomes of status assessments are altered?
2. Will pressure – state relationships between operational state indicators and the specific pressures affecting them be modified?

In both instances, this could result in a change to the management response. Should resources to support this research not be available, the alternative could be to adopt a more precautionary approach to management. Again this could result in alteration of the management response to take account of this increased uncertainty (Figure 8.1).

8.3 Examples of potential surveillance indicators

The ‘Decision’ document already specifies indicators for Descriptors 1, 3, 4 and 6 that cannot be made operational due to ill-defined pressure – state relationships or difficulties with target setting. Some of these indicators could fulfil a surveillance indicator role.

Indicators of distributional range and distribution pattern within the range have been proposed for Descriptor 1 “Biological diversity is maintained”. The distribution of demersal fish stocks can shift with ocean warming (Perry *et al.*, 2005). Such climate-induced changes in distribution may have fitness implications (DeYoung and Rose, 1993), especially for stocks that are heavily exploited around the edge of their range, e.g. Celtic Sea cod (*Gadus morhua*), which is the most southerly stock of this species and exists at an environmental frontier that might be expected to move north as mean temperature rises. Currently no operational indicators have been defined under MSFD Criterion 1.1 “Species distribution”, and setting targets for candidate indicators has proved problematic (Greenstreet *et al.*, 2012; Rindorf and Lewy, 2012). However, distributional range indicators may have a role as surveillance indicators that provide information on ecologically significant changes in distribution that can trigger research, e.g. on the desirability of, and appropriate tools for, spatial management (see section 6). Such surveillance indicators could contribute to our understanding of ecological mechanisms and pressure – state relationships underlying operational indicators for other Criteria of GES within Descriptor 1, such as “population size”, and possibly for other descriptors. Many ‘generalized interactions’ between Descriptor 1 and other MSFD Descriptors have been identified (Cochrane *et al.*, 2010), and frequently represent high-level ecosystem processes. For example, stocks which have changed distribution significantly and effectively abandoned part of their former range may not be able to reach reference levels for GES of stock size estimated based on the historic distribution area (ICES WKBALT 2013). Similarly, stocks with expanding distribution areas will likely exhibit increases in reference levels (e.g. anchovy in the North Sea). In these cases, a surveillance indicator can be used in identifying appropriate management decisions despite a lack of clear pressure-state relationship.

For Descriptor 3 each of the three criteria may include indicators of population state for which no reference levels are available. For criteria 3.1 and 3.2 so-called secondary indicators were proposed based on survey data (i.e. not stock assessments) and for criterion 3.3, i.e. healthy age and size distribution, three aspects were distinguished for which the indicators still need to be developed and agreed upon. The process towards this consists of:

1. an agreed procedure for the calculation of each selected indicator. Without this the indicator cannot be considered for subsequent steps in the further process;
2. The identification of the appropriate reference level(s) for the indicator. If there is no scientific basis for setting any reference levels the indicator will be considered a secondary indicator.

Criterion 3.3 of Descriptor 3, which concerns commercially exploited fish and shellfish, states that exploited fish stocks should exhibit 'a population age and size distribution that is indicative of a healthy stock'. At present four metrics have been considered as indicators to meet this MSFD need, but how to develop these indicators to function as operational indicators has not been obvious. However, these indicators do provide a way of tracking change. The revised manual for Descriptor 3 suggests that 'data collection for these indicators should be maintained for the time being, but not used in evaluating GES' (ICES 2015a). This language suggests that these indicators might be used to fulfil a surveillance indicator role. Further, the secondary indicators suggested for use in D3 fit exactly the definition of surveillance indicators (ICES WKMSFD3-II 2015 - ICES 2015a).

In an approach very similar to the one we suggest here for surveillance indicators, Probst *et al.*, (2013) suggest using the 95% confidence limits for entire secondary indicator time-series as reference bounds in lieu of management targets derived using more conventional methods (Link, 2005; Jennings, 2005). These secondary indicators could then support a more formal assessment process and Probst *et al.*, (2013) argue that these secondary indicators could then be used to determine whether GES had been achieved. Rather than implying that surveillance indicators could be used in much the same way as operational indicators, supporting formal assessment and informing management responses, we suggest that process illustrated by Probst *et al.*, (2013) provides a mean by which surveillance indicator might be promoted to fully operational indicators.

Ocean warming is thought to have effects on growth performance, fecundity and recruitment in marine fish (Pörtner *et al.*, 2001). The relevant pressure – state relationships are typically complex and target setting is not appropriate because upper and lower bounds may be difficult to estimate. There are various surveillance indicators that could track such climate/environment-induced change, and potentially provide information to inform stock assessments and provide diagnostic information when assessments identify significant and directional changes in population dynamics. Some examples are considered below.

Body condition in pelagic fish stocks has been considered a proxy for reproductive fitness (Koops *et al.*, 2004) or 'well-being' (Lambert and Dutil, 1997) for some fish. Fish condition metrics can provide information on ecosystem conditions, for example food limitation. Shephard *et al.*, (2014) propose an indicator that describes the average 'weight anomaly' for a pelagic fish community in a given year, i.e. deviation around an observed long-term mean. Long-term trends in condition are frequently seen in pelagic stocks and are often ignored because there is no anthropogenic pressure-state relationship or state reference point. Ignoring condition change may lead to bias in estimated spawning-stock biomass (SSB) and total allowable catches (TAC). Further, predator condition may depend on prey condition, leading to effects on other indicators (Wanless *et al.*, 2005). Fish condition could act as a surveillance indicator providing a warning sign that could trigger earlier management action than would otherwise be the case. In this instance, the primary management response may be (i)

additional research to clarify relationships between fish condition and environment, and (ii) changes to fisheries management reference points (e.g. decreased condition should trigger a more precautionary MSY and subsequent TAC reduction).

Growth rate of demersal fish can also provide insight into broader ecosystem changes such as availability and quality of prey. Changes in food availability can reflect broad environmental change (Kirby *et al.*, 2007), or indirect anthropogenic pressures, e.g. bottom trawling (Shephard *et al.*, 2010) or eutrophication (Eby *et al.*, 2005), on benthic prey communities. Climate and fishing effects on the benthos may even be synergistic (Kirby *et al.*, 2009). For example, Baltic Sea cod currently display poor body condition and minimal growth presumably as a result of a combination of deteriorating environmental conditions (eutrophication, a direct anthropogenic pressure) and declining food availability (an indirect pressure) (Eero *et al.*, 2012). Due to these changes, ICES concluded in 2014 that the stock could not be assessed using the previously used methods, but should be classified it as 'data-limited'. Growth rate and condition will continue to be monitored. Both eastern and western Baltic Cod stocks have operational indicators for the stock (SSB) and fishing pressure (F), but the growth rate surveillance indicator has added important diagnostic information which demonstrates that the decrease in stock status is partly caused by poor growth. This led to a change in stock management to increase monitoring and become more precautionary, although it may have multiple and complex pressure – state relationships.

Change in environmental conditions is strongly linked to recruitment success in many small pelagic fish (Corten, 1986; 1990; Cury and Roy, 1989; Turrell *et al.*, 1996; Ottersen and Loeng, 2000; Badalamenti *et al.*, 2002). Stock assessments and predictions for a few of these species include environment – recruitment relationships (Myers, 1998), but these relationships often disappear with environmental regime shifts (McClatchie *et al.*, 2010). A similar mechanism has been observed for scallops *Pecten maximus* in the Irish Sea, where a prolonged positive trend in sea temperature has been related to strengthening recruitment (Shephard *et al.*, 2010). A series of positive or negative deviations from stock–recruitment relationship across a number of species in an ecosystem is sometimes seen, indicating a need to be more or less optimistic about the general recruitment levels, e.g. concurrent low recruitment success of herring and saithe in the North Sea (ICES WKMSYREF3 2014). There is an obvious role for surveillance indicators in these examples, where additional information can directly explain life-history changes of direct relevance to stock assessments and fisheries management.

The potential for Descriptor 4 "Foodwebs" metrics to perform a surveillance indicator role has already been highlighted in the example used to expand the APSR management framework to incorporate a surveillance indicator function. The revised manual for D4 suggests that 'many foodweb indicators have weak or indirect links to human pressure and may show substantial variation due to factors not related to anthropogenic activities. With such indicators, it is difficult or impossible to identify values of the indicator that are desirable or undesirable in relation to human impacts' (ICES 2015b). For some foodweb indicators, there may be strong pressure-state relationships, e.g. biomass in the piscivore trophic guild is probably closely linked to fishing mortality. In contrast, the effect of fishing on biomass in the benthic guild may be indirect and time-lagged. Few if any foodweb indicators are therefore fully operational. These points are all made in the example of foodweb indicators used above to extend the APSR management framework to incorporate surveillance indicators, but it is useful to note that future revisions to the 'Decision' may well follow similar lines.

Descriptor 6 “Seafloor integrity” consists of two main criteria: 6.1. Damage to the seafloor, having regard to both pressure(s) on, and sensitivity of, habitats; and 6.2. Structural and functional condition of benthic community. As 6.1 essentially is supposed to reflect the pressure on the seafloor and the impact it has on the habitat there is scope to identify reference levels and hence come up with OI. For the 6.2 criterion, however, this is not likely and here surveillance indicators may be appropriate. Such indicators may reveal changes affecting the harvest components of the foodweb, changes caused by e.g. the landing obligation and changes caused by the introduction of new gear (e.g. pulse fishing). Climate change can induce shifts in sea temperature and ocean acidification. These changes can affect benthic invertebrate species and communities that contribute to Descriptor 6 “Seafloor integrity”. Reduced exchange of nutrients between surface waters and deeper waters is one possible impact, which could reduce the amount of organic matter sinking down to seafloor communities. This could affect the composition, functioning and biomass of deep-sea communities (ICES 2015c). Surveillance indicators that can track such climate-induced effects on the benthos could provide valuable information on broad-scale changes in ecosystem state and vulnerability, providing additional information to supplement the use of operational indicators for Descriptor 6 used within an APSR management framework.

8.4 Criteria for selecting effective surveillance indicators

Moving towards an ecosystem approach to management quickly focused attention on the need to monitor change in different attributes of a range of ecosystem components (Link, 2005; Rice, 2005), stimulating the development of numerous ‘ecological indicators’ (Link, 2005; Daan *et al.*, 2005; Shannon *et al.*, 2010; Shin and Shannon, 2010; Shin *et al.*, 2010a; Borja *et al.*, 2011; Greenstreet *et al.*, 2012b). Proliferation in the number of indicators potentially available quickly led to the introduction of criteria to assess the performance of indicators and inform the selection of indicators that would be most useful within a management context (ICES, 2000; Rice and Rochet, 2005; Rochet and Rice, 2005; Piet *et al.*, 2008). It is vital that the rationale for selecting surveillance indicators is just as clearly stated and robust as that used for selecting operational indicators. The purpose of surveillance indicators is not to monitor everything that can be monitored, but to acquire a second class of indicators that have a valuable and specific role to play in the way that operational indicators are used to implement the MSFD. Jennings and Le Quesne (2012) suggest that as the number of indicators increases, so their usefulness diminishes; the risk of between indicator inconsistency increases and it quickly becomes almost impossible to match management measures to each indicator. They stress the need for rigorous selection procedure to identify the minimum number of indicators necessary to support management. If such a statement was pertinent to operational indicators, it is even more pertinent to the selection of surveillance indicators because of the more ‘relaxed’ function that they fulfil in supporting operational indicator based APSR management frameworks. The procedure for selecting surveillance indicators therefore needs to be just as rigorous and transparent the processes used to select operational state indicators.

A formal framework for surveillance indicators, such as the one depicted in Figure 1, may employ indicators with no direct anthropogenic pressure – state relationship. It may also provide a mechanism for retaining proposed operational indicators that have adequate pressure – state relationship(s), but cannot yet be made operational because lack of knowledge and/or historical data precludes robust target setting. Even for those indicators having recognized pressure – state relationships, desirable

states may occur between upper and lower extremes, and so setting upper limits to pressure only may be inadequate. Departure from within acceptable bounds (Figure 8.2) would trigger a policy reaction (Figure 8.1). Thus the key difference between operational and surveillance indicators is relaxation of the need for surveillance indicators:

- to have clearly defined and well understood pressure – state relationships;
- to be sensitive to specific anthropogenic pressure;
- and to have established targets.

The 16 criteria for selecting operational state indicators that have been proposed by Greenstreet et al. (Submitted), WGBIODIV (2013, 2014) and WGECO (2012, 2013, 2014) were modified to reflect the principal differences between operational and surveillance indicators (Table 8.1).

Table 8.1. Criteria for selecting OSPAR Common Indicators (Greenstreet *et al.*, Submitted) interpreted for surveillance indicators (SI).

Category	Characteristic	Criterion
Type of Indicator	State or pressure	SI must track state in an ecosystem attribute that is of scientific or management interest. Specific relationships to operational indicators and APSR frameworks should be defined.
Data quality	Existing and ongoing data	Suitable data time-series must be available to determine SI. SI do not need data to support or calculate GES reference points, but time-series should be useful for identifying upper and lower bounds that may represent acceptable state.
Data quality	Indicators should be tangible	SI should be easily and accurately determined using feasible and quality assured methods. They should monitor meaningful ecosystem attributes.
Data quality	Quantitative vs. qualitative	Quantitative measurements are preferred over qualitative, categorical measurements, which in turn are preferred over expert opinions and professional judgments.
Data quality	Relevant spatial coverage	Spatial coverage can be related to corresponding operational indicator, e.g. a fish population range.
Data quality	Reflects ecosystem change caused by variation in manageable pressure(s)	SI may track change in ecosystem components having known pressure - state relationships, but having defined pressure - state relationships is not a prerequisite to SI selection.
Management	Relevant to MSFD management targets	SI will not track state in relation to targets and is not expected to comprise part of GES assessment.
Management	Relevant to management measures	SI will not trigger management of pressure, but will trigger a specified policy reaction.
Management	Comprehensible	Indicators should be interpretable in a way that is easily understandable by policy-makers and other non-scientists. The purpose of the surveillance indicator should be easy to communicate.
Management	Established indicator	Preferred surveillance indicators will have a record of use, e.g. body condition of fish. Strong scientific evidence to support the use of SI should be on record.
Management	Cost-effectiveness	Sampling, measuring, processing, analysing indicator data, and reporting assessment outcomes, should make effective use of limited financial resources.
Management	Early warning	SI that can signal potential future change in an ecosystem attribute are advantageous.
Conceptual	Scientific credibility	Scientific, peer-reviewed findings should underpin the assertion that the SI provides a true representation of variation in the ecosystem attribute in question.
Conceptual	Metrics relevance to MSFD indicator	SI need not relate directly to MSFD criteria.
Conceptual	Cross-application	SI that are applicable to more than one MSFD criterion are preferable.
Indicator suites	Indicator correlation	SI can be correlated to provide a holistic picture of possible high-level ecosystem change.

8.5 Conclusions and Discussion

Operational indicators for the MSFD are expected to have clear pressure – state relationships and a predictable response to management action. However, ecosystem interactions are frequently complex, and observed pressure – state relationships may be non-linear and/or mediated through additional (unmonitored) variables. Anthropogenic pressures may also be multiple and interacting such that cumulative or synergistic responses could make the use of operational indicators with APSR frameworks much more difficult (Crain *et al.*, 2008). Defined surveillance indicators may capture aspects of such mechanisms and provide complementary information to aid the understanding of the processes underlying changes in ecosystem state and in associated indicators. In some cases, surveillance indicators may act as a warning signal by highlighting unprecedented or directional state shifts in ecosystem components that lack operational indicators because pressure - state relationships are inadequately understood and/or target setting is not feasible. Comparing a suite of correlated surveillance indicators could help assess the likely ecosystem-scale significance of observed changes in state. Evaluating changes in surveillance indicators may have to be partly qualitative as in some cases there will be no empirical way to estimate acceptable bounds because indicator time-series are too short to capture historical acceptable states that can act as a reference range. Nevertheless, even in these circumstances, the information provided by surveillance indicators would be a useful supplement to the ‘expert opinion’ process that would have to be used in these circumstances.

An important issue in the establishment of surveillance indicators is a transparent and robust set of selection criteria that justify the incorporation of any given surveillance indicator into the APSR framework. We interpret the criteria used to select OSPAR common indicators and produce a simpler process that captures the nature and purpose of surveillance indicators. The main differences between the selection criteria for operational and surveillance indicators reflects the lack of necessity for either clear pressure – state relationships or management targets in the use of surveillance indicators.

Surveillance indicators are often influenced by a combination of several factors including climatic conditions, changes in other ecosystem components and anthropogenic pressures. To ensure that the limits continue to be relevant, they must be reevaluated if the ecosystem has exhibited pronounced regime shifts to ensure that the limit level should reflect the current regime rather than historic regimes.

Some managers and policy groups have expressed the concern that including surveillance indicators as formal MSFD indicators could increase the probability of “failing GES” assessment outcomes. In practice, the surveillance indicators can be defined as not having a direct influence on GES assessment. Indeed, change in any surveillance indicator is unlikely to be related directly to any available pressure ‘lever’. Instead, prolonged directional change in a surveillance indicator provides the option to initiate research that can identify mechanisms and clarify pressure – state relationships for operational indicators. Surveillance indicators can thus inform advice to policymakers. Surveillance indicators can also demonstrate when the system has changed and existing reference levels become less certain. It is then necessary to re-evaluate reference points to re-assess certainty. Interim adjustment of management reference points may be necessary to ensure a precautionary approach is maintained.

Resistance to the use of surveillance indicators has also been associated with uncertainty as to how managers and policy group might make use of the information provided by them; surveillance indicators had no place in indicator-based management frameworks. Our adaptation of the APSR management framework addresses this issue, providing a clear role for surveillance indicators within a management context.

8.6 References

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9 Methods applicable to providing advice for sensitive species

Sensitive species are a key component of ecosystems and form an important component of biodiversity (MSFD Descriptor 1). An indicator of sensitive fish has been developed by the Working Group on Biodiversity Science (WGBIODIV) and taken up by OSPAR, and this led the group to a general review of the appropriate methods to identify species sensitive to pressures, the estimation of exposure and the combination of these aspects in a risk based advice framework.

9.1 Existing management objectives and legislative frameworks for sensitive species

Sensitive species are potentially covered by several legislative decisions, not all of which provide the same definition of appropriate management. With regards to fishing activity, a sensitive species may also be a harvested species. Harvested species are referred to both in the CFP and the MSFD, with the CFP article 2(2) stating:

'The CFP shall apply the precautionary approach to fisheries management, and shall aim to ensure that exploitation of living marine biological resources restores and maintains populations of harvested species above levels which can produce the maximum sustainable yield.'

'In order to reach the objective of progressively restoring and maintaining populations of fish stocks above biomass levels capable of producing maximum sustainable yield, the maximum sustainable yield exploitation rate shall be achieved by 2015 where possible and, on a progressive, incremental basis at the latest by 2020 for all stocks.'

This requires that management of harvested species is based on the MSY approach rather than risk-based methods.

The Marine Strategy Framework Directive provides the legal imperative to promote ecosystem-based management across seas under the jurisdiction of the European Union (EC, 2008). The MSFD, requires Member States (MSs) of the European Union to achieve good environmental status (GES) for eleven Descriptors of GES by 2020. The EC 'Decision' document was intended to support MSs in their efforts to implement the MSFD. For Descriptor 1 "Biological diversity is maintained", the 'Decision' document suggests the use of indicators at three levels: species, habitat and ecosystem (community), (EC, 2010). At the species level, the 'Decision' document suggests that MSs use indicators of "Population abundance and/or biomass, as appropriate" (1.2.1) to monitor change in the status of Criterion 1.2 "Population Size". MSs therefore have to identify suites of species for which indicators of population abundance/biomass will be required so that temporal variation in their population size can be monitored.

MSFD Descriptor 3: Commercially exploited fish and shellfish also covers harvested species and implements the MSY approach, but here commercially exploited species have operationally been defined as species contributing more than e.g. 1% to the total landings (ICES ADGMSFD, 2015). Hence, under the MSFD, harvested species with very low catches will be addressed under Descriptor 1: Biodiversity rather than Descriptor 3. Biodiversity is not addressed using MSY principles as this seems inappropriate to the non-harvested stocks covered. As a result there is a potential conflict with the requirements for harvested species set out in the Common Fisheries Policy (CFP).

9.2 Risk based advice

Traditionally used stock assessment methods and advice such as MSY based considerations is often not applicable or informative for species that do not occur often in sampled catches and surveys. Instead, the risk posed by specified pressures to such species has been addressed using various risk-based approaches, including Productivity Susceptibility Analysis (PSA), Ecological Risk Assessments (ERAs), climate change vulnerability assessments (CCVAs) and Vulnerability to Projected Warming Assessment (VPWA) (e.g. Hobday *et al.*, 2006, IPCC, 2007; Marmaug *et al.*, 2013, Stortini *et al.*, 2015). Such approaches attempt to evaluate the risk of a species or stock to pressures based on its biological sensitivity or productivity, and its exposure to the pressure operating over its distributional range. Ecological risk assessments entail an estimate of sensitivity of specific species together with their exposure, with the combined effect of the two determining the risk level of the species in question (ICES WGECO, 2012).

9.3 Definitions

Sensitivity to a pressure is defined here as the degree to which a species or population may be negatively impacted by exposure to a particular pressure. This includes the potential of a species or population to recover from adverse effects and/or adapt to new conditions or ways of life and is a property of the inherent biological traits expressed by the species.

9.4 Methods to define sensitive species

The criteria used to determine sensitivity will differ between pressures, as species sensitive to, e.g. mortality through extraction, may differ from those sensitive to toxic substances or eutrophication (Table 9.1). Further, the pressure may affect a range of aspects including recruitment, mortality, capacity to recover (intrinsic population growth rate), resilience, etc.

Table 9.1. Pressures and likely traits of species sensitive to this pressure.

General pressure	Specific pressure	Examples of sensitive traits
Biological disturbance	Microbial pathogens	Degree of lethality of pathogens
Biological disturbance	Non-indigenous species	Competitiveness, vulnerability to predation
Biological disturbance	Selective extraction of species: mortality	Longevity, late maturation, low recolonization rate, low fecundity, low growth rate
Biological disturbance	Selective extraction of species: removal of prey	High dietary dependence on removed prey
Enrichment	Nitrogen and Phosphorus enrichment, Input of organic matter	Tolerance to hypoxia
Physical loss	Smothering, Substrate Loss	Suspension-feeders
	Death or injury by collision	Large-bodied, surface dwelling, sessile invertebrate species with stalked or complex morphology and fragile bodies (no shell or exoskeleton) and lifespans > 10 years with benthic or brooding eggs
Physical damage	Abrasion	Large-bodied, surface dwelling, sessile invertebrate species with stalked or complex morphology and fragile bodies (no shell or exoskeleton) and lifespans > 10 years with benthic or brooding eggs
Physical damage	Selective Extraction (non-living) resources	Species dependent on extracted sediments, e.g. for spawning or attachment
Physical damage	Changes in siltation	Sessile organisms and suspension-feeders; burrowing infauna
Interference with hydrological processes	Emergence regime change, pH changes, Water flow rate changes, Thermal regime changes, Salinity regime changes, Change in wave exposure	Species with high oxygen requirements, species with long pelagic life stages, species close to their distribution borders, Intertidal organisms, stenohaline or stenothermal, calcareous body parts
Other physical disturbance	Barrier to species movement	Species with fixed migration paths
Contamination	Synthetic compounds, Non-synthetic compounds, Radionuclides	Species high in the food chain, or that are bio-concentrators e.g. filter-feeders
Other physical disturbance	Marine Litter	Vulnerability to ingestion of litter or entanglement in litter
Other physical disturbance	Underwater noise	Vulnerability to sound e.g. cetaceans
Other physical disturbance	Electromagnetic changes	Sensitive to EMF e.g. some elasmobranchs

Having identified the pressures to which sensitivity is to be evaluated, we can turn to identifying traits linked to sensitivity. Examining just one or two biological or habitat-based traits is unlikely to provide a reliable indicator for sensitivity and sensitive species are generally defined based on a portfolio of traits which are related to sensi-

tivity (Davies *et al.*, 2004). Literature examples of traits showing sensitivity to four different pressures are seen in tables 9.2, 9.3 and 9.4.

Once the portfolio of traits is determined for all observed species, a sensitivity rank is estimated for each species. This can be done by examining the distribution of the trait across species (species ranked as e.g. the 10th highest out of 30, Greenstreet *et al.*, 2012) or by ranking each trait based on expert judgement (e.g. low risk (0) to high risk (5), Stortini *et al.*, 2015). Note that sensitivity may be conditioned on the pressure exerted by other sources, e.g. sensitivity to contaminants may increase sensitivity to disease.

Once the sensitivity of the species has been ranked for each trait, the total sensitivity score of the species can be derived by estimating the summed rank across traits, if necessary weighting the different traits scores according to importance (Hiddink *et al.*, 2007) or confidence in the rating (Table 9.2, Stortini *et al.*, 2015). Weighing using ranking of species according to expert judgement has advantages when the species assemblage includes traits for which the values are rarely considered high risk whereas rating according to the distribution of the trait across species has the advantage of providing an objective and consistent ranking system. From the overall species sensitivity, the 25% or 33% with the highest sensitivity, the species constituting the top 25% of all ranks summed or the species with a sensitivity higher than some threshold level can be defined as sensitive. The setting of the actual level is somewhat arbitrary but can be an emerging property of the sensitivity scores.

Table 9.2. IPCC (Intergovernmental Panel on Climate Change) confidence rankings (IPCC, 2005):

Level of confidence	Low	Medium	High	Very high
Chance	2 out of 10	5 out of 10	8 out of 10	9 out of 10
Score	0.2	0.5	0.8	0.9

9.4.1 Examples of defining sensitivity to specific pressures

The traits which are indicative of high sensitivity vary between pressures. This section provides examples of such indicator traits.

9.4.1.1 Sensitivity to selective extraction of species: mortality

Species with “slow-type” life-history traits (large-bodied, slow growing, late age and large size at first maturity, low fecundity, etc.) are particularly sensitive to the additional mortality associated with fishing activity (Jennings *et al.*, 1998; Gislason *et al.*, 2008; Hobday *et al.*, 2011; Le Quesne and Jennings, 2012). Many elasmobranch species in particular are characterized by “slow-type” traits and populations of many elasmobranchs in the North Sea had declined markedly by the 1970s (Frisk *et al.*, 2001; Greenstreet and Hall, 1996; Greenstreet *et al.*, 1999; Walker and Hislop, 1998; van Strien *et al.*, 2009). Teleost species with similar life histories also declined (Philippart, 1998; Rijnsdorp *et al.*, 1996). Life-history trait composition among the demersal assemblage as a whole had become “faster” by the 1960s (Jennings *et al.*, 1999; Greenstreet and Rogers, 2000; 2006; Greenstreet *et al.*, 2012a). The demersal fish community of the North Sea has clearly been adversely affected by fishing, and species with “slow-type” life-history traits have been the most badly affected. Promoting their recovery, and increase in the populations of these “sensitive” species, clearly therefore constitutes a step towards achieving GES.

Life-history trait data are widely available from a number of sources (Jennings, *et al.*, 1998; Jennings *et al.*, 1999; Gislason *et al.*, 2008) and the FishBase website

(www.fishbase.org) also provides data for numerous species. To derive “sensitivity” index values for all 119 of the demersal fish species recorded in the international bottom-trawl survey (IBTS) carried out in the North Sea in the first quarter (Q1) of each year 1983 to 2008 (Greenstreet *et al.*, 2011, 2012a and 2012b provide further details of the Q1 IBTS), information on the two von Bertalanffy growth equation parameters, ultimate body length (L_∞) and the growth parameter (K), and length- (L_{mat}) and age-at-first-maturity (A_{mat}) were compiled for each species. Where parameter values were not available from the sources listed above, they were estimated using the following procedures.

The *FishBase* website does provide a value for the maximum recorded length (L_{max}) of all 119 demersal fish species in the Q1 IBTS dataset. For species where L_∞ was not known, this parameter was estimated as a function of L_{max} using linear regression applied to data for species where values for both variables were available (Figure 9.1). The two von Bertalanffy variables are correlated (Gislason *et al.*, 2008), so a second regression model was developed to estimate K from L_∞ (Figure 1). The parameters derived from this second model fell well within the 95% confidence limits given for the equivalent relationship by Gislason *et al.*, (2008). Gislason *et al.*, (2008) also estimate length-at-first-maturity (L_{mat}) as a function of von Bertalanffy ultimate body length (L_∞). Linear regression models to estimate length-at-first-maturity (L_{mat}) for species where this information was missing were developed using both maximum recorded length (L_{max}) and ultimate body length (L_∞) as the explanatory variable; the (L_∞) model was the more reliable (Figure 9.2).

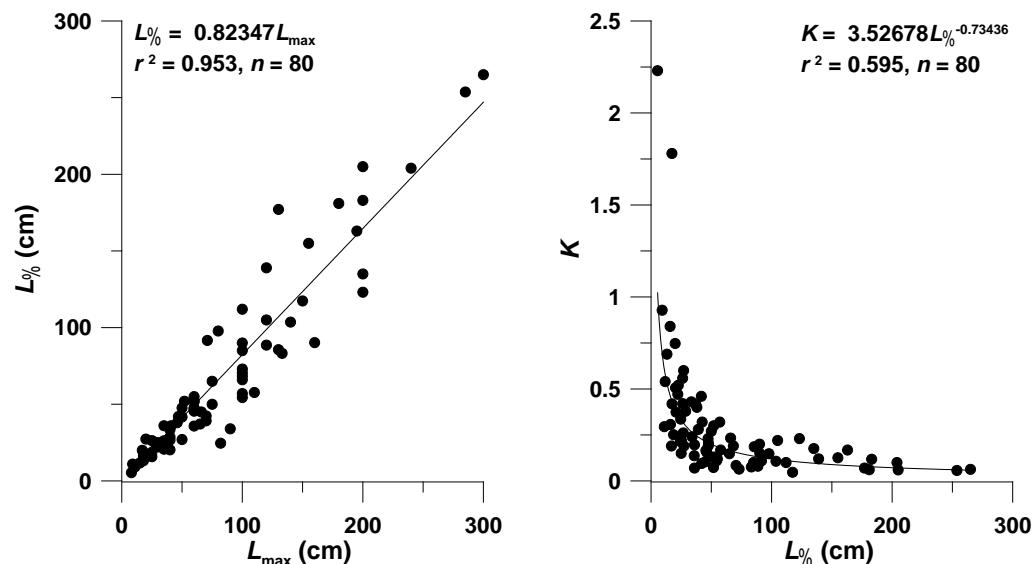


Figure 9.1. Linear regression models used to estimate ultimate body length (L_∞) from available data on maximum recorded body length (L_{max}), and to estimate the von Bertalanffy growth parameter (K) from ultimate body length (L_∞), for species where no information on the two von Bertalanffy growth equation parameters was available.

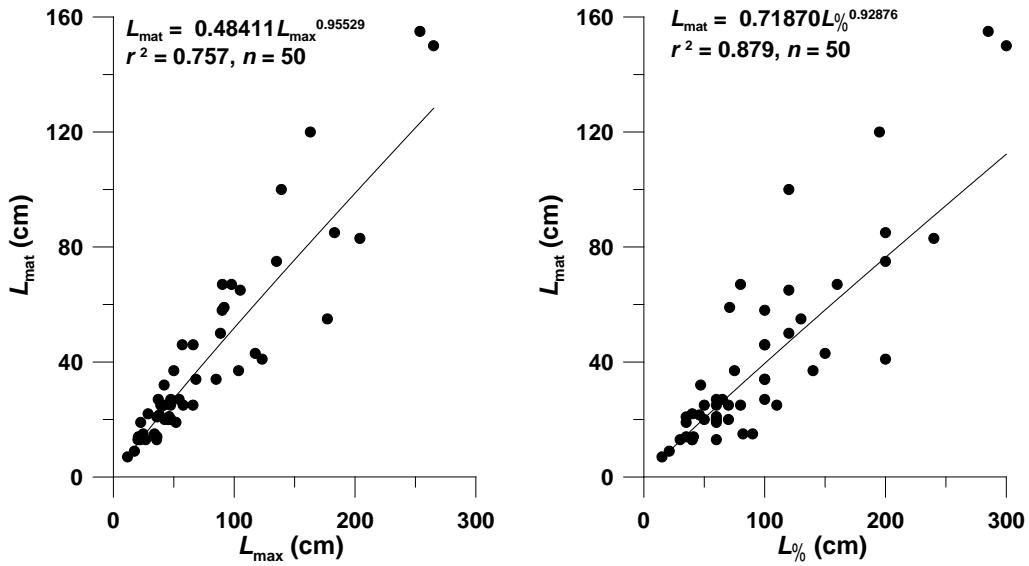


Figure 9.2. Linear regression models used to estimate length-at-first-maturity (L_{mat}) as functions of maximum recorded body length (L_{max}) and von Bertalanffy ultimate body length ($L\%$).

For most fish species, the incremental increase in length in successive time-steps follows the von Bertalanffy growth equation

$$L_{s,t} = L_{\infty,s}(1 - e^{-K_s(t-t_0)}),$$

For each focal species (s), length at a given time (t , in years) is determined by two species-specific parameters, $L_{\infty,s}$ and K_s . These are the two parameters considered as life-history trait variables in the paragraph above. Since t is relative to t_0 , setting t_0 to zero allows the equation to be rearranged to find solutions for t at specified body lengths for any particular species,

$$t = -\frac{\ln\left(1 - \frac{L_{s,t}}{L_{\infty,s}}\right)}{K_s} \quad (1)$$

Substituting estimates for length-at-first-maturity (L_{mat}) for $L_{s,t}$ then provides estimates of age-at-first-maturity (A_{mat}). However, these estimates of age-at-first-maturity also include the period of larval and post-larval development up to metamorphosis (a consequence of setting t_0 to zero). Since this growth phase is not well characterized by the von Bertalanffy growth equation, this could distort the age-at-first-maturity estimates. Gislason *et al.*, (2008) assume that fish generally go through their metamorphosis stage at a length of around 4 cm (L_{met}). This length was appropriate to the fish that they modelled, but is too large for some of the smaller fish species sampled by the Q1 IBTS. Values of $L_{\text{met}}=4$ cm were therefore assumed for all species where $L\% \geq 15$ cm, but where $L\% < 15$ cm, shorter lengths at metamorphosis were assumed: thus, if $L\% < 6$ cm then $L_{\text{met}}=1.5$ cm; if $L\% = 6$ to < 8 cm then $L_{\text{met}}=2.0$ cm; if $L\% = 8$ to < 10 cm then $L_{\text{met}}=2.5$ cm; and where $L\% = 10$ to < 15 cm then $L_{\text{met}}=3.0$ cm. Substituting these L_{met} values into equation 1 provided estimates of the age of each species at metamorphosis. Subtracting these from the initial estimates of age-at-first-maturity provided more reliable estimates for each species that excluded the duration of the uncertain larval stage. Estimates of age-at-first-maturity derived in this way were found to be a close approximation to actual values for 51 species where reported age-

at-first-maturity data were available (Figure 9.3). The linear regression obtained from this comparison was used to provide a final adjustment to the modelled estimates, to make the modelled estimates of age-at-first-maturity as similar as possible to those obtained from the literature.

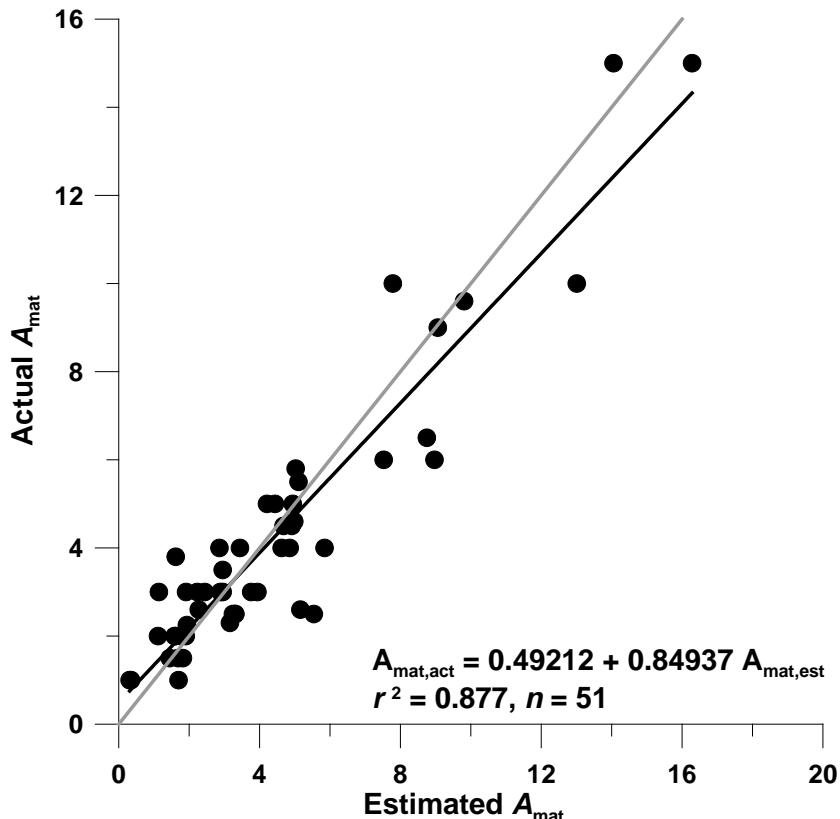


Figure 9.3. Comparison of estimates of age-at-first-maturity ($A_{\text{mat},\text{est}}$) derived from substituting estimates of length-at-first-maturity and length-at-metamorphosis into the von Bertalanffy growth equation rearranged to provide solutions for time t and specified lengths (see text for details) with actual values of age-at-first-maturity ($A_{\text{mat},\text{act}}$) provided in the literature or from *FishBase*. Grey line indicates the line of perfect prediction. Black line shows the fitted regression used to adjust the modelled estimates.

For L_∞ , L_{mat} , and A_{mat} , increasing values corresponded to increased sensitivity, but for K the reverse was true; increased sensitivity was associated with decreasing values of K . By taking reciprocals of K the direction of this relationship was reversed, ensuring that variation in all four life-history trait variables was positively related to sensitivity. Having determined values for each of the four life-history trait variables for all 119 species, these values were square-root transformed to reduce the spread in each set of data and to distribute the values more evenly within their ranges. The data for each variable were then standardized following

$$\Omega_{\text{std},s} = \frac{\Omega_s - \Omega_{\min}}{\Omega_{\text{range}}} \quad (2)$$

where Ω_s is the value for any of the four life-history trait variables for a particular species, Ω_{\min} is the minimum value for that life-history variable observed among the 119 species and Ω_{range} is the range of values for that life-history variable observed across all 119 species. $\Omega_{\text{std},s}$ is the standardized value for that life-history trait variable for the species in question. A single sensitivity index for each of the 119 species was

then derived simply by determining the arithmetic mean of the four standardized life-history trait variables. This index ranged between 0 representing the least sensitive, or most resilient, species and 1 representing the most sensitive species. Values for each of the life-history trait variables, and the resulting sensitivity index, are given for all 119 species in Table 1. The forty species in the upper 33 percentile and forty species in the lower 33 percentile were categorized as “sensitive” species and “resilient” species respectively (Table 9.3).

Table 9.3. List of 119 species recorded in the Q1 IBTS and likely to occur in the Greater North Sea giving species-specific values for four life-history traits: the two von Bertalanffy growth equation parameters, ultimate body-length (L_∞) and growth rate (K), and length- (L_{mat}) and age-at-first-maturity (A_{mat}). Bold font indicates variable values obtained from the literature and *FishBase* website; other values were estimated using various modelling approaches (see text for details). Maximum recorded length (L_{max}) values for each species obtained from *FishBase* are also given as this was a key variable used to derive estimates for missing life-history trait data. The resultant sensitivity indices for each species, and each species classification to a sensitivity category, are provided.

Scientific Name	Common Name	L_{max}	L_∞	K	L_{mat}	A_{mat}	Sensitivity Index	Sensitivity Category
<i>Aphia minuta</i>	Transparent goby	8	5.4	2.230	3.4	1.0	0.011	Resilient
<i>Crystallogobius linearis</i>	Crystal goby	5	4.1	1.247	2.7	0.9	0.011	Resilient
<i>Diplecogaster bimaculata</i>	Two-spotted clingfish	6	4.9	1.091	3.2	1.0	0.024	Resilient
<i>Pomatoschistus pictus</i>	Painted goby	6	4.9	1.091	3.2	1.0	0.024	Resilient
<i>Taurulus lilljeborgi</i>	Norway bullhead	7.4	6.1	0.935	3.9	1.0	0.035	Resilient
<i>Pomatoschistus minutus</i>	Sand goby	11	9.2	0.928	5.6	1.1	0.049	Resilient
<i>Spinachia spinachia</i>	Fifteen spined stickleback	22	17.3	1.780	10.1	1.0	0.058	Resilient
<i>Callionymus reticulatus</i>	Reticulated dragonet	11	9.1	0.699	5.6	1.3	0.064	Resilient
<i>Liparis montagui</i>	Montagu's sea snail	12	9.9	0.656	6.0	1.3	0.072	Resilient
<i>Phrynorhombus norvegicus</i>	Norwegian topknot	12	9.9	0.656	6.0	1.3	0.072	Resilient
<i>Arnoglossus laterna</i>	Scaldfish	25	15.8	0.840	9.3	1.1	0.075	Resilient
<i>Lesueurigobius friesii</i>	Fries's goby	13	10.7	0.618	6.5	1.3	0.076	Resilient
<i>Centrolabrus exoletus</i>	Small-mouthed wrasse	18	13.2	0.689	7.9	1.3	0.080	Resilient
<i>Artediellus europaeus</i>	Atlantic hook-ear sculpin	15	12.4	0.557	7.4	1.5	0.090	Resilient
<i>Echiichthys vipera</i>	Lesser weever	15	12.4	0.557	7.4	1.5	0.090	Resilient
<i>Liparis liparis</i>	Sea snail	15	12.4	0.557	7.4	1.5	0.090	Resilient
<i>Syngnathus rostellatus</i>	Nilsson's pipefish	17	20.0	0.747	11.6	1.2	0.095	Resilient
<i>Callionymus maculatus</i>	Spotted dragonet	16	13.2	0.531	7.9	1.5	0.097	Resilient

Scientific Name	Common Name	L_{max}	L_∞	K	L_{mat}	A_{mat}	Sensitivity Index	Sensitivity Category
<i>Ciliata mustela</i>	Five-bearded rockling	25	20.6	0.383	11.9	1.0	0.111	Resilient
<i>Ciliata septentrionalis</i>	Northern rockling	20	16.5	0.451	9.7	1.6	0.116	Resilient
<i>Leptoclinus maculatus</i>	Spotted snake blenny	20	16.5	0.451	9.7	1.6	0.116	Resilient
<i>Lycenchelys sarsi</i>	Sar's wolf eel	20	16.5	0.451	9.7	1.6	0.116	Resilient
<i>Triglops murrayi</i>	Moustache sculpin	20	16.5	0.451	9.7	1.6	0.116	Resilient
<i>Zeugopterus regius</i>	Eckstrom's topknot	20	16.5	0.451	9.7	1.6	0.116	Resilient
<i>Leptagonus decagonus</i>	Atlantic poacher	21	17.3	0.435	10.1	1.7	0.122	Resilient
<i>Callionymus lyra</i>	Dragonet	30	22.2	0.471	13.0	1.5	0.125	Resilient
<i>Gaidropsarus mediterraneus</i>	Shore rockling	50	27.0	0.600	15.3	1.5	0.126	Resilient
<i>Agonus cataphractus</i>	Hooknose	21	17.4	0.419	9.0	2.0	0.129	Resilient
<i>Syngnathus typhle</i>	Broad-nosed pipefish	35	26.2	0.558	14.9	1.5	0.129	Resilient
<i>Buglossidium luteum</i>	Solenette	15	11.7	0.540	7.0	3.0	0.131	Resilient
<i>Pomatoschistus microps</i>	Common goby	9	11.0	0.295	6.7	2.3	0.134	Resilient
<i>Trisopterus minutus</i>	Poor cod	40	20.3	0.506	13.0	2.0	0.136	Resilient
<i>Arnoglossus imperialis</i>	Imperial scaldfish	25	20.6	0.383	11.9	1.9	0.142	Resilient
<i>Zeugopterus punctatus</i>	Topknot	25	20.6	0.383	11.9	1.9	0.142	Resilient
<i>Ctenolabrus rupestris</i>	Coldsinny wrasse	18	16.1	0.307	9.5	2.2	0.146	Resilient
<i>Pholis gunnellus</i>	Butterfish	25	26.3	0.420	15.0	2.0	0.154	Resilient
<i>Raniceps raninus</i>	Tadpole fish	28	23.1	0.352	13.3	2.1	0.156	Resilient
<i>Trisopterus esmarkii</i>	Norway pout	35	22.6	0.520	19.0	2.3	0.158	Resilient
<i>Mullus surmuletus</i>	Striped red mullet	40	33.4	0.430	18.7	1.9	0.164	Resilient
<i>Capros aper</i>	Boarfish	30	24.7	0.335	14.1	2.2	0.165	Resilient
<i>Echiodon drummondii</i>	Pearlfish	30	24.7	0.335	14.1	2.2	0.165	Intermediate
<i>Parablennius gattorugine</i>	Tompot blenny	30	24.7	0.335	14.1	2.2	0.165	Intermediate
<i>Trisopterus luscus</i>	Bib	46	38.0	0.400	21.6	1.5	0.165	Intermediate
<i>Merlangius merlangus</i>	Whiting	70	42.4	0.320	20.0	1.5	0.176	Intermediate

Scientific Name	Common Name	L_{\max}	L_{∞}	K	L_{mat}	A_{mat}	Sensitivity Index	Sensitivity Category
<i>Microchirus variegatus</i>	Thickback sole	35	20.7	0.374	14.0	3.0	0.176	Intermediate
<i>Taurulus bubalis</i>	Sea scorpion	25	18.7	0.251	10.9	2.6	0.176	Intermediate
<i>Hippoglossoides platessoides</i>	Long rough dab	82	24.6	0.336	15.0	2.6	0.177	Intermediate
<i>Limanda limanda</i>	Common dab	40	26.7	0.261	13.0	2.3	0.178	Intermediate
<i>Chelidonichthys cuculus</i>	Red gurnard	50	41.7	0.460	25.0	2.0	0.184	Intermediate
<i>Myoxocephalus scorpius</i>	Bullrout	90	34.0	0.240	15.0	2.0	0.187	Intermediate
<i>Gobius niger</i>	Black goby	18	16.9	0.190	9.9	3.2	0.200	Intermediate
<i>Syphodus melops</i>	Corkwing wrasse	28	24.4	0.214	14.0	3.2	0.210	Intermediate
<i>Gobius cobitis</i>	Giant goby	27	25.3	0.217	14.4	3.1	0.211	Intermediate
<i>Solea vulgaris</i>	Dover sole	70	39.2	0.280	25.0	2.5	0.217	Intermediate
<i>Pegusa lascaris</i>	Sand sole	40	28.7	0.379	22.0	4.0	0.222	Intermediate
<i>Enchelyopus cimbrius</i>	Four-bearded rockling	41	36.0	0.196	14.0	3.0	0.224	Intermediate
<i>Spondylisoma cantharus</i>	Black sea bream	60	52.0	0.300	28.2	2.5	0.229	Intermediate
<i>Triglops pingelii</i>	Ribbed sculpin	20	27.3	0.190	15.5	3.5	0.232	Intermediate
<i>Pagellus erythrinus</i>	Pandora	60	35.8	0.137	13.0	2.6	0.233	Intermediate
<i>Microstomus kitt</i>	Lemon sole	65	37.1	0.415	27.0	4.0	0.235	Intermediate
<i>Lampetra fluviatilis</i>	European river lamprey	50	41.2	0.230	22.7	3.1	0.238	Intermediate
<i>Syngnathus acus</i>	Great pipefish	50	41.2	0.230	22.7	3.1	0.238	Intermediate
<i>Lumpenus lampretaeformis</i>	Snake blenny	50	47.6	0.205	20.0	3.0	0.243	Intermediate
<i>Lycodes vahlii</i>	Vahl's eelpout	52	42.8	0.223	23.5	3.2	0.244	Intermediate
<i>Platichthys flesus</i>	Flounder	60	47.3	0.230	25.0	3.0	0.246	Intermediate
<i>Eutrigla gurnardus</i>	Grey gurnard	60	46.2	0.156	21.0	2.5	0.248	Intermediate
<i>Trachinus draco</i>	Greater weever	53	43.6	0.220	24.0	3.2	0.248	Intermediate
<i>Glyptocephalus cynoglossus</i>	Witch	60	45.5	0.165	20.0	3.0	0.254	Intermediate
<i>Mullus barbatus</i>	Red mullet	30	25.0	0.150	14.3	4.3	0.257	Intermediate
<i>Gaidropsarus vulgaris</i>	Three-bearded rockling	60	47.5	0.191	27.0	3.0	0.259	Intermediate
<i>Entelurus aequoreus</i>	Snake pipefish	60	49.4	0.201	26.9	3.5	0.268	Intermediate

Scientific Name	Common Name	L_{max}	L_∞	K	L_{mat}	A_{mat}	Sensitivity Index	Sensitivity Category
<i>Etomopterus spinax</i>	Velvet belly	60	49.4	0.201	26.9	3.5	0.268	Intermediate
<i>Myxine glutinosa</i>	Hagfish	80	65.9	0.163	25.0	2.7	0.271	Intermediate
<i>Melanogrammus aeglefinus</i>	Haddock	100	68.3	0.190	34.0	2.5	0.273	Intermediate
<i>Phycis blennoides</i>	Greater forkbeard	110	57.7	0.168	25.0	3.5	0.283	Intermediate
<i>Scophthalmus rhombus</i>	Brill	75	50.0	0.270	37.0	4.5	0.289	Intermediate
<i>Pleuronectes platessa</i>	Plaice	100	54.4	0.110	27.0	2.5	0.290	Intermediate
<i>Psetta maxima</i>	Turbot	100	57.0	0.320	46.0	4.5	0.298	Intermediate
<i>Amblyraja radiata</i>	Starry ray	100	66.0	0.233	46.0	4.0	0.310	Intermediate
<i>Lepidorhombus whiffiagoni</i>	Megrim	60	51.8	0.073	19.0	2.6	0.311	Sensitive
<i>Cyclopterus lumpus</i>	Lumpsucker	60	55.0	0.120	29.7	4.0	0.322	Sensitive
<i>Pollachius pollachius</i>	Pollack	130	85.6	0.186	44.8	3.7	0.327	Sensitive
<i>Chelidonichthys lucerna</i>	Tub gurnard	75	65.0	0.148	34.7	4.5	0.331	Sensitive
<i>Zoarces viviparus</i>	Viviparous blenny	52	52.0	0.130	28.2	5.1	0.332	Sensitive
<i>Gadus morhua</i>	Cod	200	123.1	0.230	41.0	3.8	0.333	Sensitive
<i>Merluccius merluccius</i>	Hake	140	103.6	0.107	37.0	3.0	0.350	Sensitive
<i>Labrus bergylta</i>	Ballan wrasse	66	45.0	0.104	24.7	6.2	0.356	Sensitive
<i>Lophius budegassa</i>	Black bellied angler	100	85.0	0.100	34.0	4.0	0.362	Sensitive
<i>Scyliorhinus canicula</i>	Lesser spotted dogfish	100	90.0	0.200	58.0	5.0	0.367	Sensitive
<i>Raja clavata</i>	Thornback ray	120	105.0	0.220	65.0	5.0	0.377	Sensitive
<i>Leucoraja circularis</i>	Sandy ray	120	98.8	0.121	51.2	5.3	0.402	Sensitive
<i>Leucoraja fullonica</i>	Shagreen ray	120	98.8	0.121	51.2	5.3	0.402	Sensitive
<i>Petromyzon marinus</i>	Sea lamprey	120	98.8	0.121	51.2	5.3	0.402	Sensitive
<i>Lophius piscatorius</i>	Angler	200	135.0	0.176	75.0	5.0	0.415	Sensitive
<i>Raja montagui</i>	Spotted ray	80	97.8	0.148	67.0	6.0	0.416	Sensitive
<i>Squalus acanthias</i>	Spurdog	160	90.2	0.150	67.0	6.5	0.419	Sensitive
<i>Sebastes viviparus</i>	Norway haddock	35	36.0	0.070	20.9	9.6	0.425	Sensitive
<i>Brama brama</i>	Ray's bream	100	70.5	0.084	37.4	7.6	0.433	Sensitive

Scientific Name	Common Name	L_{max}	L_∞	K	L_{mat}	A_{mat}	Sensitivity Index	Sensitivity Category
<i>Mustelus asterias</i>	Starry smooth hound	140	115.3	0.108	59.1	5.9	0.438	Sensitive
<i>Raja undulata</i>	Undulate ray	100	112.0	0.100	57.5	6.3	0.448	Sensitive
<i>Molva dypterygia</i>	Blue ling	155	155.0	0.126	77.8	5.0	0.450	Sensitive
<i>Chimaera monstrosa</i>	Rabbit ratfish	150	123.5	0.103	63.0	6.1	0.454	Sensitive
<i>Anguilla anguilla</i>	European eel	133	83.2	0.076	43.6	8.2	0.468	Sensitive
<i>Leucoraja naevus</i>	Cuckoo ray	71	91.6	0.109	59.0	9.0	0.471	Sensitive
<i>Pollachius virens</i>	Saithe	130	177.1	0.070	55.0	4.6	0.477	Sensitive
<i>Molva molva</i>	Ling	200	183.0	0.118	85.0	5.5	0.482	Sensitive
<i>Helicolenus dactylopterus</i>	Bluemouth	47	42.0	0.095	32.0	15.0	0.484	Sensitive
<i>Scyliorhinus stellaris</i>	Nurse hound	170	140.0	0.094	70.8	6.6	0.486	Sensitive
<i>Brosme brosme</i>	Torsk	120	88.6	0.080	50.0	9.6	0.492	Sensitive
<i>Sebastes marinus</i>	Redfish (marinus)	100	73.0	0.064	38.6	9.7	0.492	Sensitive
<i>Anarhichas lupus</i>	Catfish	150	117.4	0.047	43.0	6.0	0.502	Sensitive
<i>Hippoglossus hippoglossus</i>	Halibut	240	204.0	0.100	83.0	5.8	0.506	Sensitive
<i>Raja brachyura</i>	Blond ray	120	139.0	0.120	100.0	9.3	0.531	Sensitive
<i>Galeorhinus galeus</i>	Tope	195	163.0	0.168	120.0	10.0	0.544	Sensitive
<i>Anarhichas minor</i>	Spotted catfish	180	181.0	0.061	89.8	9.7	0.605	Sensitive
<i>Mustelus mustelus</i>	Smooth hound	200	205.0	0.060	100.8	9.8	0.627	Sensitive
<i>Conger conger</i>	Conger eel	300	265.0	0.063	150.0	10.0	0.683	Sensitive
<i>Dipturus batis</i>	Skate	285	253.7	0.057	155.0	15.0	0.751	Sensitive
<i>Somniosus microcephalus</i>	Greenland shark	730	601.1	0.032	273.9	16.4	1.000	Sensitive

Sensitivity was highly correlated to each of the sensitive traits, and hence, if it is only possibly to collect information for some of the sensitive traits, focusing on a few can be attempted. Using length at maturity or L_∞ only to predict sensitivity results in the relationships given in Figure 9.4, explaining 89 and 87% of the variation, respectively. Using only one of these parameter to identify sensitive species (sensitivity of 0.311 corresponding to either length=0.3085 cm or L_∞ =60.3 cm) led to about 14% misses (sensitive but not recorded as such), 9% false responses (not sensitive but recorded as such) and 86% correct classifications (Table 9.4).

Table 9.4. Classification quality using length at maturity or L_∞ only to predict sensitivity. A total of 40 species out of 119 were defined as sensitive in the full analysis.

	L_∞	L_{mat}
Correct	34	35
Miss	6	5
False	3	4

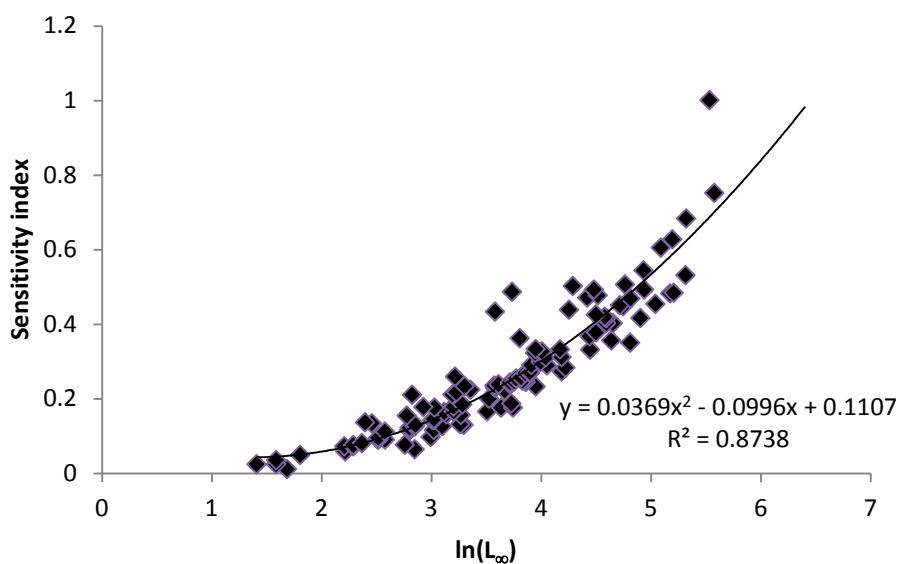
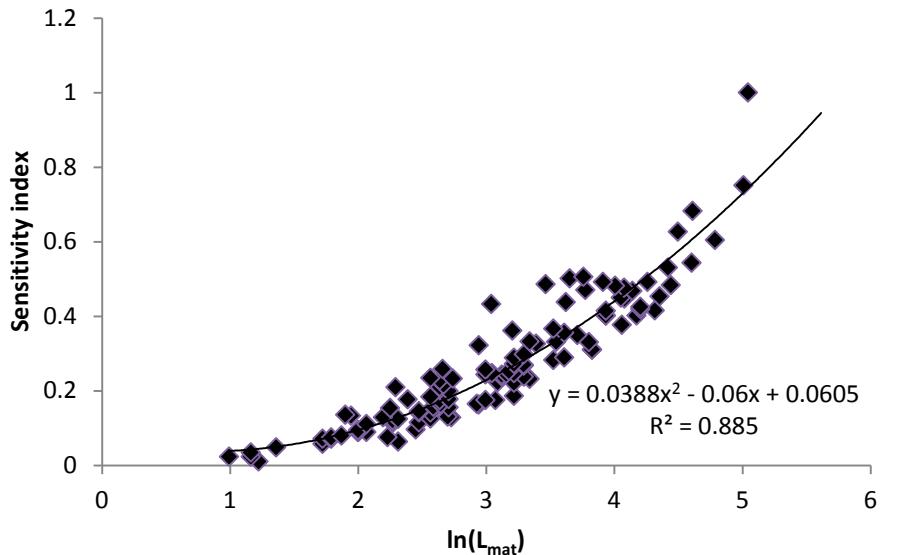


Figure 9.4. Relationship between length at maturity and L_∞ and sensitivity. Lines are 2nd degree polynomials.

The approach was used to identify a suite of species deemed to be sensitive to the additional mortality associated with fishing activity in the North Sea (Greenstreet *et al.*, 2012b). Trends-based targets for recovery (Jennings and Dulvy, 2005; Link, 2005; Shin *et al.*, 2005) were subsequently proposed for each species in this suite. A further target, based on demonstrating a significant departure from the binomial distribution, was then established for the number of species within the suite that would be

required to meet their own individual species-specific trends-based targets for population recovery. Meeting this final over-arching target would support the conclusion that significant progress towards achieving GES was being made with regard to these depleted “sensitive” species (Greenstreet *et al.*, 2012b). However, some species among the full suite of 40 “sensitive” species in the North Sea were so rarely encountered by the Q1 IBTS that the data available were considered inadequate to support this level of analytical assessment. In order to be considered in the assessment therefore, a ‘filtering’ criterion was conceived: species should be recorded in at least half of the years of the survey time-series.

Application of this ‘filtering’ criterion reduced the number of “sensitive” species actually assessed from the full suite of 40 to only 27. Thirteen of perhaps the rarest and potentially most threatened species in the community were not assessed, which might be of concern to stakeholders involved in marine conservation. However examination of the life-history trait composition of the 27 adequately sampled species and 13 inadequately sampled species revealed no significant difference. Life-history trait composition of the 27 assessed species was therefore representative of the life-history trait composition of the full suite of 40 “sensitive” species. This implies that, should a significant number of the 27 assessed species meet their trends-based population-recovery targets, then population recoveries are also likely to be occurring among the 13 non-assessed species. Under such circumstances, it is probable that recovering species, whose population abundance is increasing, will be sampled by the survey with increasing frequency and, at some point therefore, meet the sampling adequacy ‘filtering’ criterion. Following successful remedial management that initiates a general recovery situation in “sensitive” species populations, the number of species capable of supporting analytical assessment is likely to increase over time (Greenstreet *et al.*, 2012b).

The indicator proposed by Greenstreet *et al.*, (2012b) has been adopted as the “common indicator” to be used by MSs sharing MSFD subregions within the regional seas area covered by OSPAR to fulfil the EC ‘Decision’ indicator 1.2.1 “Population abundance and/or biomass” role. The , indicator derivation, target setting process and assessment analysis carried out by Greenstreet *et al.*, (2012b) constitutes the formal testing this indicator for the Greater North Sea subregion, but similar formal testing in other subregions, such as the Celtic Seas and the Bay of Biscay and Iberian Coast, has still to be undertaken. The procedure for defining suites of “sensitive” species presented here is generic; it can be applied to any groundfish survey dataset to define suites of “sensitive” species-specific to each survey. Both the Celtic Seas and Bay of Biscay and Iberian Coast MSFD subregions are adequately covered by appropriate groundfish surveys (e.g. Sánchez and Serrano, 2003; Shephard *et al.*, 2011; Modica *et al.*, 2014). Defining suites of “sensitive” species for these two additional MSFD subregions is therefore a feasible prospect.

9.4.1.2 Sensitivity to selective extraction of species: removal of prey

Dickey-Collas *et al.*, (2014) used a range of traits to determine the sensitivity of predators to removal of specific prey. The traits included a distributional contribution, a diet composition contribution and a contribution based on observed effects of food shortage. The distributional contribution was defined as the summed effect of whether the species was globally scarce (1) or not (0) and whether a significant proportion of the population resided in the North Sea (1) or not (0). The diet contribution was defined as 0 if the proportion in the diet was low (<20%), 1 if the proportion was medium (20–50%), and 2 if the proportion was high (>50%). Documented effects on

growth or reproduction resulted in a scoring of either 0 (no effects documented), 1 (effects of food abundance on growth documented), or 2 (effect of FF abundance on reproductive output documented) for recorded impacts. The total scoring was the sum of scores obtained from distributional aspects, dietary aspects, and documentation of effects, except when the dietary aspect score was 0, in which case the distributional score was also set to 0. Species with a total score of 4 or above were concluded to be highly vulnerable (Table 9.5).

Table 9.5. Species sensitivity to shortage of commercially exploited forage fish in the North Sea.
Intensity of grey shading indicates sensitivity.

Predator Name	Distribution in North Sea (local or widespread)	Distributional sensitivity	Total forage fish in diet	Reported effects of low forage fish abundance?	Total Sensitivity
Minke Whale <i>Baleonoptera acutorostrata</i>	Widespread	Medium	62%	None	3
Grey Seal <i>Halichoerus grypus</i>	Local	Medium	42%	Likely	3
Common Seal <i>Phoca vitulina</i>	Local	Medium	49%	Low	2
Harbour Porpoise <i>Phocoena phocoena</i>	Widespread	Medium	6%	Low	0
Striped dolphin <i>Stenella Coeruleoalba</i>	Widespread	Low	16%	None	0
Great skua <i>Catharacta skua</i>	Local during breeding	High	10-95%	High	6
European shag <i>Phalacrocorax aristotelis</i>	Local	Medium	98%	High	5
Sandwich tern <i>Sterna sandvicensis</i>	Local	Low	99%	High	4
Common guillemot <i>Uria aalge</i>	Local during breeding	Low	70%	High	4
Razorbill <i>Alca torda</i>	Widespread	Low	68%	Likely	3
Black-legged kittiwake <i>Rissa tridactyla</i>	Local during breeding	Low	36%	Yes	3
Atlantic puffin <i>Fratercula arctica</i>	Widespread	Low	71%	None	2
Northern gannet <i>Morus bassanus</i>	Widespread	Low	34%	None	1
Lesser black-backed gull <i>Larus fuscus</i>	Widespread	Medium	7-25%	None	0
Northern Fulmar <i>Fulmarus glacialis</i>	Widespread	Low	15%	None	0

Predator Name	Distribution in North Sea (local or widespread)	Distributional sensitivity	Total forage fish in diet	Reported effects of low forage fish abundance?	Total Sensitivity
Whiting <i>Merlangius merlangus</i>	Widespread, substock structure	Low	30%	Likely	2
Cod <i>Gadus morhua</i>	Widespread, substock structure	Low	21%	Likely	2
Saithe <i>Pollachius virens</i>	Widespread	Low	41%	None	1
Horse mackerel <i>Trachurus tharracurus</i>	Widespread	Low	34%	None	1
Starry ray <i>Amblyraja radiata</i>	Widespread		24%	None	1
Grey Gurnard <i>Eutrigla gurnardus</i>	Widespread	Low	23%	None	1
Haddock <i>Melanogrammus aeglefinus</i>	Widespread, substock structure	Low	20%	Low	0

9.4.1.3 Sensitivity to physical damage

Bottom fishing activities exert pressure by causing death or injury to benthic invertebrates through physical damage caused by direct contact (collision) with the fishing gear. Recently, biological traits expressed by organisms have been used to assess changes in benthic communities due to various pressures (Bremner *et al.*, 2003; 2006). When conducting such analyses it is important to include a large number of traits in order to adequately characterize the communities (Bremner *et al.*, 2003; 2006). The EU FP7 project BENTHIS created a table of 12 biological traits and modalities for use in biological traits analysis (Table 3.2). Some of these traits have been shown to respond to fishing pressure.

For each of the traits identified in the BENTHIS study, we ranked the modalities according to their sensitivity to the pressure ‘Physical Damage’ based on expert opinion (Table 9.6). All of the traits had relevance to this analysis except for Bioturbation and Feed mode which were considered to be a function and so not assessed. From this ranking the most sensitive species were those that are large (greater than 100 mm) stalked, tubed, reef- or bed-forming sessile epifaunal with fragile or unprotected bodies. These traits make them vulnerable to direct mortality and are indicated by the assignation of (1) in the ranking of 1–3 in Table 9.6 under the Instantaneous category. Not all species that are affected will have long recovery times which factor into the calculation of sensitivity. We considered the mobility potential and the longevity of the species as well as its egg and larval reproductive traits to factor into recovery potential and similarly ranked traits from 1 (slow recovery) to 3 (more rapid recovery) in Table 9.6. Species that are long-lived (>10 years) will also have varying degrees of sensitivity which may be reflected in their egg care strategies. Species with benthic or brooded eggs may take longer to recover if recolonization is required from greater distances than species with pelagic larvae. Therefore they may be considered more sensitive than those that have pelagic eggs and are ranked accordingly. In the current analysis, this was assumed though the reverse may be true in some cases. From these we produced a score for impact and recovery which when summed provide a rank-

ing of species sensitivity to the trait where smaller values convey higher sensitivity. In applying this ranking to some benthic species found in European waters the reef-building coral *Lophelia pertusa*, the habitat forming sponge *Geodia barretti*, and the bed-forming flat oyster *Ostrea edulis* all readily emerge as highly sensitive species, with scores between 10 and 14 (Table 9.7). In contrast the thick shelled bivalve *Astarte* spp. and the errant polychaete *Phyllodoce* spp. both had scores greater than 24. These species are likely at the extremes of the scoring range which gives a range of 14 points within which to evaluate responses. With more data it may be possible to see bimodal responses with some species clearly being sensitive and others not.

Table 9.6 Traits of species of benthos ranked according to sensitivity to pressure causing death or injury by collision.

Trait	Categories (infauna and epifauna)	Mechanism	Rank Category	Rationale for Ranking
Bed/reef formers	Reef-builder (brReef)	Instantaneous	1	High ranking due to larger size through aggregation
Bed/reef formers	Bed-former (brBed)	Instantaneous	1	High ranking due to larger size through aggregation
Living habit	Tube-dwelling (lhTube)	Instantaneous	1	Sessile organisms less able to avoid impact
Living habit	Attached to bed (lhAtt)	Instantaneous	1	Sessile organisms less able to avoid impact
Morphology (epifauna)	Stalked/pen-shaped (mPen)	Instantaneous	1	Upright organisms more vulnerable to gear interaction
Morphology (epifauna)	Stalked/fan-shaped (mFan)	Instantaneous	1	Upright organisms more vulnerable to gear interaction
Morphology (epifauna)	Stalked/complex (mComp)	Instantaneous	1	Upright organisms more vulnerable to gear interaction
Protection	Fragile (pFrag)	Instantaneous	1	Unprotected species more susceptible to damage
Protection	Unprotected (pUnprot)	Instantaneous	1	Unprotected species more susceptible to damage
Sediment position	Surface (spSurf)	Instantaneous	1	Position vulnerable to gear interactions
Size range (mm)	>500 (s500)	Instantaneous	1	Larger sized organisms are more likely to be in direct contact with gear
Size range (mm)	101–200 (s101–200)	Instantaneous	1	Larger sized organisms are more likely to be in direct contact with gear
Size range (mm)	200–500 (s200–500)	Instantaneous	1	Larger sized organisms are more likely to be in direct contact with gear
Living habit	Free living (lhFree)	Instantaneous	2	Some ability to escape interaction
Living habit	Crevice/under stone (lhCrev)	Instantaneous	2	Some ability to escape interaction
Protection	Protected (skin/exoskeleton) (pExo)	Instantaneous	2	Some ability to escape interaction
Sediment position	0–5 cm (spInf_0to5)	Instantaneous	2	Some ability to escape interaction
Size range (mm)	21–100 (s21–100)	Instantaneous	2	Some ability to escape interaction
Bed/reef formers	None (brNone)	Instantaneous	3	Impact not linked to this trait
Living habit	Burrow-dwelling (lhBurrow)	Instantaneous	3	Dependent on position in the sediment which is covered elsewhere
Living habit	Epi/endo zoic/phytic (lhEpi)	Instantaneous	3	Dependent on characteristics of host organisms which are covered elsewhere

Trait	Categories (infauna and epifauna)	Mechanism	Rank Category	Rationale for Ranking
Morphology (epifauna)	Round-bodied (mRound)	Instantaneous	3	Morphology may be less susceptible to damage by non-crushing impacts
Morphology (epifauna)	Flat/encrusting (mFlat)	Instantaneous	3	Morphology less susceptible to damage
Protection	Protect (robust shell) (pRob)	Instantaneous	3	Morphology confers protection while intact
Sediment position	5–10 cm (spInf_6to10)	Instantaneous	3	Position less vulnerable to gear interactions
Sediment position	>10 cm (spInf_10)	Instantaneous	3	Position less vulnerable to gear interactions
Size range (mm)	≤10 (s10)	Instantaneous	3	Size less vulnerable to gear interactions
Size range (mm)	11–20 (s11–20)	Instantaneous	3	Size less vulnerable to gear interactions
Egg development location	Eggs – benthic (edSex_ben)	Recovery	1	Direct impact may prevent or delay local recovery
Egg development location	Eggs – brooded (edSex_br)	Recovery	1	Direct impact on adult may prevent or delay local recovery through limited local dispersal
Larval development strategy	Direct (ldD)	Recovery	1	Limited dispersal potential
Longevity	>10 (l10)	Recovery	1	May be slow growing with episodic recruitment
Mobility	Sessile (mobSess)	Recovery	1	No adult migration
Mobility	Burrow (mobBur)	Recovery	1	Limited adult migration
Egg development location	Asexual/fragmentation (edAsex)	Recovery	2	May recolonize from damaged fragments in some species
Larval development strategy	Lecithotrophic (ldLc)	Recovery	2	May have greater larval survivorship
Longevity	1–2 (l1to2)	Recovery	2	R-selected species with high turnover times
Longevity	3–10 (l3to10)	Recovery	2	Mostly R-selected species with high turnover times
Mobility	Crawl (mobCrawl)	Recovery	2	Limited ability to move into the area from adjacent areas
Egg development location	Eggs – pelagic (edSex_pel)	Recovery	3	May recolonate from undisturbed populations
Larval development strategy	Planktotrophic (ldPk)	Recovery	3	May recolonate from undisturbed populations
Longevity	<1 (l1)	Recovery	3	R-selected species with high turnover times
Mobility	Swim (mobSwim)	Recovery	3	May recolonate through adult migration from undisturbed populations

Trait	Categories (infauna and epifauna)	Mechanism	Rank Category	Rationale for Ranking
Feeding mode	Suspension (fSusp)	Function		Direct relationship with ecosystem functioning
Feeding mode	Surface deposit (fSurf)	Function		Direct relationship with ecosystem functioning
Feeding mode	Subsurface deposit (fSub)	Function		Direct relationship with ecosystem functioning
Feeding mode	Scavenger (fScav)	Function		Direct relationship with ecosystem functioning
Feeding mode	Predator (fPred)	Function		Direct relationship with ecosystem functioning

Table 9.7. Scoring of the Sensitivity of Selected Species Using the System Outlined in Table 9.6.

Species	Bed/re ef former s	Livin g habit	Mopholo gy (epifauna	Protecti on	Sedime nt positio n	Size rang e (mm)	Impa ct Score	Egg developme nt locatio n	Larval developme nt strategy	Longevi ty	Mobilit y	Recove ry Potenti al Score	Sensitivity Score
Lophelia pertusa	1	1	1	1	1	1	6	2.5	3	1	1	7.5	13.5
Geodia baretti	1	1	1	1	1	1	6	1	1	1	1	4	10
Ostrea edulis	1	1	1	1	1	1	6	3	3	1	1	8	14
Astarte spp.	3	2	3	3	2	3	16	3	3	1	1	8	24
Phyllodoce spp.	3	3	3	1	1	3	14	3	2	3	2	11	25

9.4.1.3.1 Sensitivity and Ecosystem Function

The previous section outlines how biological traits can be used to identify sensitive species and habitats in response to pressure caused by Physical Damage. In ToR a, we have shown how biological traits can be mapped to ecosystem functions. In this example feeding strategy has a direct effect on ecosystem function expressed through changes in community composition (Table 9.8).

Table 9.8. Example of benthic taxa with different modalities for the trait Feeding Mode.

Trait	Modality	Examples of Species groups	Reference
Feeding Mode	Suspension-feeder	Most invertebrates classed as epifaunal, such as some ciliates, sponges, corals, some coelenterates, some rotifers, barnacles, some pelecypods and gastropods, some sedentary polychaetes, brachiopods, bryozoans, phoronids, some echinoderms and tunicates including crinoids, as well as some of the infaunal animals such as pelecypods, certain gastropods and an echiuroid worm.	Bullivant (1968)
	Deposit-feeder	Some bivalve molluscs and gastropods, many polychaetes including spionids, some sea cucumbers, and some crustaceans.	Jumars (1993); Lopez <i>et al.</i> , (1989); Levinton and Kelaher (2004)
	Predator/Scavenger	Asteroid starfish, many crabs, some gastropods, isopods, gammarids, lyssianid amphipods, anemones, and nemertean worms.	This report section ToR b

Kaiser *et al.*, (2006) performed a meta-analysis of 101 different fishing impact studies and evaluated the short-term response of benthic invertebrates to fishing disturbance of different gears on different substrata with respect to 2 major functional groups: deposit- (DF) and suspension-feeders (SF). Feeding mode was responsive to different gears and habitats as indicated in Figure 9.5. Further, repeated trawling in an area attracts benthic scavengers which feed on discards and moribund animals (e.g. Ramsay *et al.*, 1998; Groenewold and Fonds, 2000; Catchpole *et al.*, 2006; Shepherd *et al.*, 2013) and may locally enhance populations (Shepherd *et al.*, 2013). Consequently, this feeding mode is also responsive to the fishing pressure and can impart changes to the benthic communities and ecosystems. These contrasting feeding strategies reflect different trophic pathways, interspecific interactions and benthic functioning (Table 9.8) and the fact that they are responsive to the pressure means that they are appropriate traits to consider in a sensitivity analysis. However this does not mean that the response to the pressure is directly associated (cause and effect) with these feeding modes. More likely position in or on the sediment and morphology were the traits that were directly affected and those traits were correlated with the examined function.

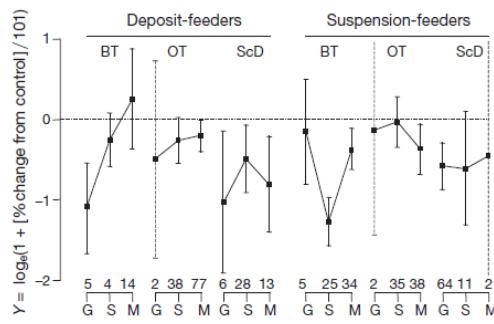


Fig. 5. Mean initial response (up to 7 d after impact), with 95 % CI, of deposit- and suspension-feeding fauna to (BT) beam-trawling, (OT) outer-trawling and (ScD) scallop-dredging in (G) gravel, (S) sand and (M) muddy sand/mud habitats combined. Dashed lines: CI where only 2 points available for mean calculation, and hence some intervals extend outside plotted range. Values above x-axis: number of data points in each mean calculation. Adequate test for a significant initial impact: whether 95 % confidence interval crosses zero-response line

Figure 9.5. Response of the trait Feeding Mode expressed by two modalities (Deposit- and Suspension-Feeders) to fishing activities (Figure 5 of Kaiser *et al.*, 2006).

9.4.1.4 Sensitivity to thermal regime changes

Stortini *et al.*, (2015) evaluated the sensitivity of 33 fish and invertebrate species, on the scale of the Scotian Shelf to ocean warming. They examined the traits latitudinal range, food specificity, habitat specificity, population status, resilience, larval dispersal and adult mobility and used a 1–5 scoring system to determine the risk for each species and trait. Further, they combined the sensitivity of different traits in a weighting procedure according to their importance as derived from the literature. The species were then defined as vulnerable when the combined effect of sensitivity and exposure exceeded a specified threshold. Only one species was identified as being at risk to a mild warming, whereas 18 species were identified as sensitive to severe warming, including a large fraction of the planktivores and medium benthivores.

9.5 Methods to define exposure

The spatial and temporal exposure can be evaluated either qualitatively based on expert judgment or quantitatively based on e.g. pressure maps. It is determined by a variety of factors, and the overall average exposure level is only one of these. Marine communities are seldom homogeneous in space; size/age structure and species composition vary as a function of environmental variables such as benthic habitat and prey community (Fraser *et al.*, 2008; Ehrich *et al.*, 2009). Spatially aggregated species are often particularly sensitive to pressures, and the current spatial and temporal distribution is likely to be the result of both habitat characteristics and historic pressures. Metrics of size-structure in the Celtic Sea suggests that the fish community can change as a function of ‘local’ fishing intensity (Shephard *et al.*, 2011; De Castro *et al.*, 2015). Fine-scale fishing impacts likely accrue because of temporal stability in the spatial distribution of effort (Hinz *et al.*, 2003; Kaiser, 2005) relative to environment and habitat characteristics (e.g. substratum, Nilsson and Ziegler, 2007). Fishing impacts on the seabed (e.g. Hinz *et al.*, 2009) and on target communities (Babcock *et al.*, 2005; Fréon *et al.*, 2005) can thus be spatially discrete rather than even across the range of a population. However in the North Sea several studies (e.g. Piet and Hintzen, 2012) have shown that fishing becomes more evenly distributed and the area not affected decreases over time due to slightly changing patterns.

Spatial effects of fishing may be most evident among species sensitive to fishing pressure such as elasmobranchs. Rogers *et al.*, (1999) note that current elasmobranch abundance is lowest in the most heavily fished (southeastern) part of the North Sea, although previously such species were common in this area (Philippart, 1998). Walker and Heessen (1996) speculate that areas in the North Sea that are difficult to access with towed gear could become *refugia* for elasmobranch populations. Correspondingly, Shephard *et al.*, (2011) found that Celtic Sea survey sampling sites that occurred in the lowest 10% of the observed fishing effort range recorded 10 species of elasmobranch including the critically endangered *Dipturus* spp.; the most intensely fished 10% of sites had only three elasmobranch species, including two IUCN listed as Least Concern. Therefore, exposure should be evaluated spatially whenever possible.

9.6 Definition of risk

Ecological risk assessments have traditionally been based on a likelihood-consequence approach for estimating the risk of a rare or unpredictable event (Williams *et al.*, 2011), but when an assessment of ongoing (current) pressure is needed, then an exposure-effect analysis is more suitable (Smith *et al.*, 2007). Such analyses can use qualitative descriptors to assess the vulnerability of habitats and assess the potential for ecosystem-based management (EBM) at subregional scale (Samhouri and Levin, 2012) and regional scales (Knights, 2015). Risk assessment can therefore play an increasingly important role in integrating science, policy and management (CENR, 1999; Piet, 2015).

In classic risk assessments, risk can be estimated as either:

$$\text{Ecological Risk} = \text{Likelihood} * \text{Consequence} \quad (1)$$

or

$$\text{Ecological Risk} = \text{Exposure} * \text{Effect} \quad (2)$$

Both consequence and effect terms reflect how much the species is affected when there is an interaction. This is clearly affected by the sensitivity of the species to that pressure but where the level of the pressure (i.e. intensity or frequency in case of fishing) is only implicit. For the pressures in the marine ecosystems according to the MSFD (Table 9.1), (2) seems more appropriate and risk is often estimated as the spatial or spatio-temporal overlap between the species and the human-induced pressure. Depending on the indicator used to determine overlap, this may or may not reflect the overall risk induced by the overall exposure level (e.g. overlap could be high but if the pressure level is low, this may not mean that risk is also high). To approach this issue, two aspects of exposure can be estimated, the overall average exposure (e.g. general fishing pressure in fleets catching the given species or similar) and the spatio-temporal exposure (e.g. the spatial co-occurrence of fishing and the given species). Alternatively, exposure to different levels of pressure can be analysed separately. These levels can be defined based on the pressure-state relationships of the sensitive species.

9.7 Advice

In general, providing advice for sensitive species is often hampered by the low abundance of the species leading to reported catches being rare and possibly misreported. However, aggregate indicators like the sensitive fish indicator proposed by Greenstreet *et al.*, (2012b) tend to be less sensitive to changes in single species caused by

sampling variation, and for these, it should be possible to provide advice. As an example, WGECO examined the possibility to provide advice on the development of state and pressure for the sensitive fish indicator.

9.7.1 Advice on sensitive fish

The advice on sensitive fish species should contain the historical development in the state of the indicator, the general pressure and the spatial overlap between exposure and sensitive species. This information will demonstrate whether:

- a) the state of the indicator is in accordance with GES;
- b) the direction of the indicator relative to GES (towards GES, away from GES, fluctuating within GES);
- c) changes in the state and direction of the indicator is related to general or spatial pressure.

If the indicator is in accordance with GES or moving towards GES at a sufficient rate, the advice may state that general pressure and spatial pressure should not increase.

If the indicator is outside GES or moving away from GES, management measures can be suggested for reducing either general pressure or spatial pressure to ensure that risk is consistent with moving towards GES.

The rebuilding of stocks of sensitive species is likely to be a three step process, and the transition between steps will likely require management decisions based on policy rather than science. Currently, the species with the highest recovery potential are increasing in many areas as a response to the reduction in general fishing pressure in many areas. As these more responsive sensitive species rebuild, it will be necessary to re-evaluate reference levels to ensure that these reflect stable population levels of these species.

However, the general decline in fishing pressure is unlikely to be sufficient for the more sensitive species. Some of these may be protected by spatial measures. Areas of consistently low pressure may have acted as *de facto refugia* (and/or as population sources), and these areas may have considerable conservation importance for species whose populations have declined overall, and which cannot sustain a population in other areas. Because *de facto* refuges have no formal protection, they are extremely vulnerable to redistribution of pressures. Several mechanisms might result in effort redistribution, e.g. adaptation to the landings obligation and/or to changes in gear capacity, e.g. electronic pulse beam trawls which can work on previously ‘unfishable’ ground. Studies of the benthos (Hughes *et al.*, 2014) suggest that even a single bottom-trawling event can have a substantial negative impact on the seafloor community. The benefit of refuges could thus be quickly negated. Ring-fence protection for known refuge areas would offer protection. Because these areas are (almost by definition) seldom fished, proposed legal protection may find relatively easy political support.

It should be recognized that sensitive species may be subject to other pressures than those caused by fishing and which may be inhibiting recovery. Many of the sensitive species are boreal, and they are therefore under combined pressure of climate and fishing. Further, species which have declined substantially historically may now be in so-called ‘mortality pits’ where their natural mortality has increased or their recruitment rate decreased to a level where population increase is insufficient to rebuild the stock even in the absence of fishing. Finally, the most highly sensitive species may be vulnerable to a fishery at any level and hence requiring their restoration may conflict

with the CFP objective to “provide conditions for economically viable and competitive fishing capture”, compromise the food provisioning service and require measures beyond what would be required for the MSFD concept of ‘under sustainable exploitation’.

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Annex 2: Agenda

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

8–15 April 2015, ICES HQ, Denmark

Agenda

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

Wednesday 8 April

- 10.00 Opening of the meeting
 - Adoption of TOR and Agenda
 - Overview of volunteers to work on different tors and overview of presentations prepared for the meeting
 - Initial discussion of TOR d, Review, develop and assess indicators of Good Environmental Status of Distribution in the context of MSFD and management response;
- 13.00 Lunch
- 14.00 Reconvene
- 16.00 Coffee
- 17.30 Develop Following Day Workplan, Updates from earlier in day, Wrap up
- 18.00 Adjourn

Thursday 9 April

- 9.00 Initial discussion of TOR c: Evaluate the ecological consequences of restoring stocks to MSY levels and the degree to which fisheries are “balanced;”
- 10.30 Coffee
- 13.00 Lunch
- 14.00 Initial discussion of TOR a: Develop and assess indicators of Good Environmental Status of the benthic community;
- 16.00 Coffee
 - Subgroups on tors a, c and d
- 17.30 Develop Following Day Workplan, Updates from earlier in day, Wrap up
- 18.00 Adjourn

Friday 10 April

- 9.00 Subgroups on tors a, c and d
10.30 Coffee
Subgroups on tors a, c and d
13.00 Lunch
14.00 Reconvene
Subgroups on tors a, c and d
16.00 Coffee
Subgroups on tors a, c and d
17.30 Develop Following Day Workplan, Updates from earlier in day, Wrap up
18.00 Adjourn

Saturday 11 April

- 9.00 Preliminary check on WG report elements
Revisit, as need be, TORs a, c and d
10.30 Coffee
13.00 Lunch
14.00 Initial discussion of TOR b: Develop indicators of scavengers, examine their relation to discard amounts and evaluate the potential effect of a landing obligation on the benthic ecosystem;
Initial discussion of TOR e: Identify data weaknesses in the data available to address WGECO ToRs and recommend priority areas for data collection and model development.
16.00 Coffee
17.30 Develop Following Day Workplan, Updates from earlier in day, Wrap up
18.00 Adjourn

Sunday 12 April

Writing day, no plenaries

Monday 13 April

- 9.00 Subgroup reporting TORs a-e
10.30 Coffee
Drafting session
13.00 Lunch
14.00 Reconvene
Drafting session

16.00 Coffee

18.00 Adjourn

Tuesday 14 April

9.00 Subgroup reporting

10.30 Coffee

Drafting session

13.00 Lunch

14.00 Reconvene

Drafting session

16.00 Coffee

18.00 Adjourn

Wednesday 15 April

9.00 Scope out next year meeting plan, schedule, TOR

10.30 Coffee

Tying up loose ends

13.00 Adjourn

Annex 3:WGECO terms of reference for the next meeting

The **Working Group on the Ecosystem Effects of Fishing Activities** (WGECO), chaired by Anna Rindorf, Denmark, will meet in Copenhagen, Denmark 6–13 April 2016 to:

- a) Develop and assess indicators of Good Environmental Status for seafloor integrity: Further develop indices of impact on the seafloor and sensitivity of the benthic community to different pressures, and the link to function (joint ToR with BEWG)
- b) Investigate possible indicators of scavengers, examine their relation to discards amounts and evaluate the spatial effect of a landing obligation on the scavengers
- c) Use the data available to evaluate the ecological consequences of restoring stocks to MSY levels and the degree to which fisheries are “balanced”.
- d) Distributional indicators in the context of MSFD: developing operational and surveillance indicators and exploring the link to pressure and drivers
- e) Review and further develop indicators of state of sensitive species throughout the ICES area and suggest risk based advice on sensitive species
- f) Consider methods to integrate indicators in support of integrated assessment of GES at the MSFD descriptor level (in collaboration with the DEVOTES project and building on work from WGBIOV)

WGECO will report by 24 April 2015 to the attention of the Advisory Committee.

ToRs for consideration for the 2017 meeting

- a) Expanding the DPSIR framework with links to ecosystem services: How does state and impact relate to the provision of ecosystem services and human wellbeing
- b) Continue the development of foodweb indicators and comment on the suggested foodweb indicators from WKFooWI and WGSAM
- c) Identify data weaknesses in the data available to address WGECO ToRs and recommend priority areas for data collection and model development

Supporting Information

Priority	The current activities of this Group will lead ICES into issues related to the ecosystem affects of fisheries, especially with regard to the application of the Precautionary Approach. Consequently, these activities are considered to have a very high priority.
Scientific justification	<p>Term of Reference a)</p> <p>WGECO considers that the development and assessment of indicators of Good Environmental Status of the benthic community is a key area of development in relation to the MSFD. With this ToR, WGECO will follow and review the progress in international projects working on this topic and work towards the development of operational objectives and indicators for the benthic community as well as an improved understanding of benthic foodweb dynamics. In 2015, WGECO will specifically review the effect of trawling on benthic community composition.</p> <p>Term of Reference b)</p> <p>The implementation of a landing obligation is likely to result in major changes to the impact of fishing on particularly benthic scavenging species. The magnitude of both direct and indirect effects are unknown at present and further work is needed in order to provide advice on the direction and magnitude of change as a result of the landing obligation.</p> <p>Term of Reference c)</p> <p>Determining the ecological consequences of restoring stocks to MSY levels and the degree to which fisheries are “balanced” ideally requires a large metadataset to ensure that analyses are based on all available data. To ensure that such data are available to WGECO, the group will issue a data call in January 2016 to obtain the data necessary to (i) examine time-trends in stocks that are rebuilding and dependent stocks; their prey and predators; compare these trends with predictions from qualitative analyses or LeMans or other quantitative multispecies models and (ii) establish the distribution of total catch among size classes (catch size spectrum), species (catch species dominance curves) and functional groups. The data call will request two types of data: 1. Stock assessment output for all stocks assessed in an area: F and SSB estimates tabulated in a consistent way. There is a trade-off between the number of stocks included and the length of the time-series. (i) needs at least 10 years, whereas (ii) might be shorter especially if it allows to include more discard estimates. (ii) may be more a snapshot that could be compared across areas. 2. (ii) may also be addressed based on catch data = landings + discard estimates. Landings are available from public databases; discard estimates for TAC-species are going to be available in the discard atlases being prepared as input to the regional discard management plans; issue is discard estimates for non-TAC species.</p> <p>Term of Reference d)</p> <p>Distributional indicators have obtained little attention in</p>

ICES in the interim between the end of WGFE and 2015, when WGECO redressed the issue. WGECO considers this an important area of research in connection with the MSFD and proposes that the work should be continued in cooperation with e.g. SGSPATIAL. The work in 2016 will focus on developing operational and surveillance indicators and exploring the link to pressure and drivers.

Term of Reference e)

Indicators of state of sensitive species have been developed for fish in the North Sea. However, there are numerous ICES areas where a list of sensitive species is not available for fish. Even less information exists for other taxa. To allow advice on sensitive species throughout the ICES area, there is a need to expand the methods used to encompass other areas and to suggest a format in which risk based advice on sensitive species can be given.

Term of Reference f)

One of the key challenges identified in the recent ICES process to provide advice on the manual for the MSFD descriptors 3, 4 and 6 was the integration across indicators of indicator state relative to reference levels. Such an integration is necessary to derive a status at descriptor level of whether or not GES is achieved. Building on work from WGBIOV, WGECO will consider methods to integrate indicators in support of integrated assessment of GES at the MSFD descriptor level in collaboration with the DEVOTES project.

Resource requirements	The research programmes which provide the main input to this group are already underway, and resources are already committed. The additional resource required to undertake additional activities in the framework of this group is negligible.
Participants	The Group is normally attended by some 20–25 members and guests.
Secretariat facilities	None.
Financial	No financial implications.
Linkages to advisory committees	There are no current direct linkages with the advisory committees.
Linkages to other committees or groups	There is a very close working relationship with the groups of the Fisheries Technology Committee, WGBIRD, BEWG, WGBIODIV and WGSAM.
Linkages to other organizations	OSPAR, HELCOM

Annex 4: Recommendations to other EGs

REQUEST	GROUP	EXPLANATION
Information on spatial distribution and abundance of key scavengers	BEWG	In 2016, WGECO will continue the work on the effect of the landing obligation on benthic communities. As part of this, WGECO is considering whether it is possible to collaborate with BEWG to obtain a list of the most abundant potential scavengers, a digital map of the biomass and/or abundance of key scavengers and the proportion of the benthic biomass which is made up by key scavengers.
Annual spatial distribution of VMS from fishing vessels using towed gear	SGVMS	In 2016, WGECO will continue the work on the effect of the landing obligation on benthic communities. As part of this, WGECO will attempt to collate a spatial map of the current high discard intensity areas. To indicate potential discard areas, WGECO suggest to use pings of vessels using towed gear from the point of the first vessel fishing activity to the end of the trip (i.e. including steaming back to port). The data can be aggregated by gear type and should cover the gear types Otter Trawl; OTB1, OTB2, Beam Trawl: TBB, and Seine SSC and SDN

Activity 1 – Investigate possible indicators for proportion of benthic biomass which are key scavengers

A term of reference will be constructed in collaboration with BEWG.

Activity 2 - Investigate possible indicators for biomass or abundance of key scavengers

A term of reference will be constructed in collaboration with BEWG.

Activity 3 – Investigate possible indicators based on important areas of spatial overlap of key scavenger species distribution and discards from main fleets

Together with BEWG and SGVMS, the abundance and spatial distribution of the key scavenger species, the general discard level and the spatial distribution of discards from the main fleets will be established, respectively. The spatial overlap of these distributions will be examined to highlight areas where key scavenger species may observe a food shortage under a landing obligation. This may lead to the development of a pressure indicator reflecting the reduction of introduced organic matter as a result of the landing obligation.