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1 Executive Summary

It has been several years since ICES has completed a comprehensive review of the effects of fishing on the North Sea ecosystem, and this year our ToR (Section 3) has allowed us to revisit this important topic. We have reviewed the impacts of each of the major gears in terms of their effects on all components of the ecosystem, and for the first time since 1995 have described the international distribution of fishing effort of beam trawls, otter trawls (including seine gears), and small-meshed fisheries throughout the North Sea. Compilation of such data at an international level, and at the scale of the ICES rectangle, was a frustrating task and was fraught with problems of data compatibility and quality. ICES will need to plan carefully if such an advisory request should come from an external customer, and in section 7 we consider ICES readiness to provide advice to the Regional Advisory Councils, and have included lessons learnt from our work on this ToR.

Last year WGECO identified the need for fully Integrated Ecosystem Assessments to link manageable human activities with the pressures they cause in the marine ecosystem. The matrix of pressures and components of the ecosystem provides a useful tool for prioritising the key interactions in the ecosystem, and a weighting system was developed based on the spatial extent (local or widespread) and intensity of the interaction (chronic or acute) (section 4). Indicators which might be appropriate for managing the human activities responsible for these pressures were identified.

In our 2005 Report we presented a detailed analysis of how ecosystem effects of fishing could be included into the provision of routine fisheries advice. In turn, SGMAS considered our proposals and Section 5 continues this dialogue. We identify ecosystem considerations that should be taken into account in an ecosystem approach to fisheries management, and suggest that these should be part of routine activities not an optional extra. WGECO feels that this would be an attitude consistent with good risk management practices applied in many other fields. Where knowledge or data are inadequate, we clarify the work that needs to be undertaken so that improvements can be made.

WGECO has advised on the development and implementation of the Ecological Quality Objective (EcoQO) approach in OSPAR for several years, and feels that ICES is now in a position to provide clear advice on a way forward with implementing the EcoQO on changes in the proportion of large fish in the fish community (section 6). This EcoQ element, as measured in research trawls, is a useful indicator of the effect of fishing, a useful state indicator for the fish community and is indicative of wider changes in the biodiversity of the ecosystem. WGECO concluded that the EcoQO can be further progressed as part of an objectives-based management framework and so has defined a goal for the fish community to 'Halt as rapidly as possible, and begin to reverse by 2010, both the decline in the mean weight and the proportion of large fish'. Large fish were defined as those greater than 30cm in length, and short, medium and long-term operational targets were suggested. In the short-term it is suggested that the decline in proportion of these fish size measures in survey catches should be halted immediately. In the medium term, targets for fish size should be based on the time necessary to restore fish populations to conditions in the early 1980s when ICES generally considered stocks to be sustainable. In the longer term, targets could be revised using improved information on the ecological consequences of an over-fished fish community and societal choices for more or less ambitious conservation objectives.

ICES have begun a dialogue with the Regional Advisory Councils (RACs) to develop an understanding of their requirements for advice and how this advice might be provided. In section 7 we provide a summary of the ecosystem effects of fishing for all components relevant to the North Sea RAC (based on work in section 3), and extend this approach to the other RACs. Although not comprehensive, it highlighted the many interactions between fisheries and ecosystems for which ICES lacks knowledge to provide quality advice. While many effects can be generalised across the region, most specific studies relate to the North Sea and Baltic Sea, and in many cases, the extent of the effect will depend on the nature and scale of the fishing activity in an area. ICES currently lacks the capacity to deal with geographically referenced data and this skill will become increasingly important as advice is requested on a range of geographically-related fisheries management measures.

This year WGECO continued to assist the Regional Ecosystem Group for the North Sea (REGNS) in their work on an Integrated Assessment (section 8). In our review of the dataset used by the working group we found a number of inconsistencies and anomalous data entries which may have affected the outcome of their analyses. We also reviewed the coverage of the ecosystem components in the database supplied to us and suggested taxa and components which have important roles in ecosystem function, and which could be used in future. Comments were also made on the analytical approach adopted by REGNS to encourage wider discussion of the methods used for Integrated Assessment and in their later thematic assessments.

2 Opening of the meeting

The Working Group on Ecosystem Effects of Fishing Activities (WGECO) met at ICES HQ, Copenhagen, from 5-12 April 2006. The list of participants and contact details are given in Annex 1.

We were welcomed to ICES on the morning of 5 April by Adi Kellerman, the ICES Director of Science Programmes, who expressed his appreciation for the work done by WGECO during its' 15 year history, and confirmed the importance of past reports in developing the assessment of fisheries effects and informing the ICES advisory process on a wide range of ecosystem issues. The Terms of Reference for the meeting were then discussed, and a plan of action was adopted with individuals allocated separate tasks to begin work on all ToR. This was followed by a joint meeting with members of the Workshop on Fisheries Management in Marine Protected Areas (WKFMMPA) where issues of shared interest were discussed, particularly the work planned by WGECO to review and report on the full effects of fishing on the North Sea ecosystem.

The Terms of Reference for the meeting are given in Annex 2:

2.1 Acknowledgements

WGECO gratefully acknowledges the contributions made by a number of individuals and groups who have provided support to the meeting.

We were allowed access to databases collated by the EU project MAFCONS and by the EU STECF (Scientific, Technical and Economic Committee for Fisheries) subgroup for the review of stocks, SGRST. Both databases provided very valuable information on fishing activities, based on voluntary contributions of the participating countries. We thank the members of MAFCONS and STECF for supporting further evaluation of their data within WGECO, without which we could not have effectively addressed our ToR. Any scientific group or individual interested in conducting further analyses on the basis of these data bases, should contact the members of MAFCONS (greenstreet@marlab.ac.uk) or the participants of the respective countries in the STECF subgroup (see http://stecf.jrc.cec.eu.int/ event.php?id=23) for permission.

We would like to thank the Chair and members of WGNSSK for supplying data and commentary on the bycatch of fish in industrial fisheries which was valuable in our interpretation of the effects of these gears on fish communities and the wider ecosystem.

Several individuals also kindly provided their support, especially Simon Northridge and Anne McLay who both gave up their time to provide help at short notice to complete our descriptions of fish effects. Finally, the Working Group would particularly like to thank Bodil Chemnitz and other members of the ICES Secretariat for their willing support to enable the meeting run smoothly and to ensure that the final report is completed to schedule.

3 TOR a) The effects of fishing on the North Sea ecosystem

Review and report on the full effects of fishing on the North Sea ecosystem, grouped according to the suite of ecosystem components identified in previous meetings and where necessary in a regional context, with an emphasis on; i) the direct effects of demersal trawling on benthic species, ii) the ecosystem effects of the small-meshed fisheries targeting fish not for human consumption, iii) the ecological consequences of discarding and iv) the indirect effects of fishery removals on community scale indicators identified as promising at past WGECO meetings.

3.1 Introduction and approach

WGECO has been considering the various effects of fishing on ecosystem dynamics since its inception in 1991, and this period has also seen a large growth in peer reviewed science dealing with this issue (e.g. Camphuysen & Garthe, 2000; Greenstreet & Rogers, 2000; Hall et al., 1993; Jennings & Reynolds, 2000; Lindeboom & de Groot, 1997) and a number of synthesis works (e.g. Hall, 1999, Kaiser & de Groot, 2000). Much of the recent literature can be regarded as adding examples and increasing the generality of the conclusion reached by documentation of effects in new geographic regions.

The direct effects of fisheries on target species, by-catch species and habitats are well characterised. However, while a range of studies, including field and modelling ones have shown the scale of the indirect effects, these are much less tractable and so more poorly known. There is some evidence for local indirect effects, including competition between fisheries and marine mammals/seabirds. For example, the breeding success of kittiwakes along the eastern coast of Scotland is lower in years when sandeel fisheries are active than in non-fishery years (Frederiksen et al, 2004; Scott *et al.* in press). However, examples of such effects are not common. Due to the long-lived / low breeding productivity characteristics of seabirds and marine mammals, responses to changes in fish populations e.g. size spectra, caused by fishing, may be delayed and prolonged.

Given this body of knowledge it seems appropriate to consider the effects of fisheries on the North Sea ecosystem through a consideration of the specifics, including the effects of the gears used in the North Sea on the various components of the ecosystem, the areas these gears are used, and the locations where these effects are most pressing. Consideration of this ToR therefore proceeds with an explicit consideration of spatial distribution of ecosystem components and impacts. It is hoped that this will provide information in a form useful to resource managers and specifically contribute to the emerging discussions on marine spatial management.

For the purposes of this ToR we have taken the North Sea to be defined by the RAC boundaries (see section 7 for further details).

3.2 Ecosystem components identified by WGECO

In 2004 WGECO developed a list (Table 3.2.1) of key ecosystem components that could be used to guide the development of management measures aimed at delivering ecosystem level objectives (ICES, 2004).

Table 3.2.1 WGECO proposed ecosystem components which should be considered in a holistic framework for ecosystem protection.

Habitats – physical and chemical attributes
Nutrients
Plankton (phytoplankton and zooplankton)
Benthos
Fish community
Commercial fish and shellfish
Marine mammals
Seabirds

WGECO recognised that this classification is artificial and primarily reflects ecological divisions. It was further noted that while commercial fish stocks are part of the fish community the information needs will differ between the various groups seeking advice and support. Their needs are therefore best served by considering the fish community and commercial fish separately.

The ecosystem components identified in Table 3.2.1 were selected to provide the minimum number of components that need to be managed for while providing adequate coverage of all the system. Habitat is taken to refer to the physical and chemical environment and hence includes water quality and the physical (substratum) aspects of the environment. Nutrients include the essential biological nutrients and consideration of their sources, fluxes and biogeochemical transformations. Plankton (phytoplankton and zooplankton) and benthos both provide food resources while the former is also environment for larval stages of benthos and fish. The benthos element also includes their role as structural habitat agents. Fish community includes the whole fish assemblage. The top predators, marine mammals and seabirds are, at least for the public, the most conspicuous elements of the marine ecosystem and are often regarded as environmental sentinels.

3.3 Fishing impacts on components by gear type

It is generally recognised that the first fishing event has proportionally more impact than subsequent ones (Collie *et al.*, 2000). However, the effects of multiple events are cumulative such that multiple fishing effects by a low impact gear may in fact exceed the changes induced by a single pass of a more impacting gear. This relationship is further complicated by the fact that in most biological systems mechanisms for recovery exist. Therefore the key issue is not the absolute frequency of an impacting activity but the frequency relative to the recovery time for that system. Thus, the impacts of fishing need to be considered in terms of intensity of impact, frequency of impact, and nature of the impacted system, in particular its ability to, and rate of recovery. The following sections describe the effects of fishing on these components grouped by the major gear types in the North Sea.

3.3.1 Beam trawling

3.3.1.1 Habitats – physical and chemical attributes & Nutrients

The effects of fishing on habitat are related to the physical disturbance by bottom gears in contact with the seafloor. In summary these include removal of large physical features, reduction in structural biota and a reduction in complexity of habitat structure (leading to increased homogeneity) (ICES 2002, 2003a). The extent of these changes is related to the types of fishing gear being used and the initial level of complexity in both physical and biogenic structure (see Auster & Langton, 1999; Johnson, 2002). Structurally complex habitats tend to offer a greater diversity of food, physical shelter from disturbances and predation and, for some species, provide features such as sites for egg laying (Lokkeberg, 2005). Much of the work that has already been undertaken in relation to alteration of habitat in the towpath has taken place in areas other than the North Sea (see review in ICES 2002, 2003a). Given that many of the habitats studied previously are of high structural complexity, we suggest that the comparability with effects in the North Sea is likely to be low.

At the same time, the resuspension of sediments that occurs during the trawling process may be associated with the release of contaminants and heavy metals that have previously been stabilised in the sediments. The effects of resuspension events on nutrient fluxes have also been studied, but again, most of the available literature is not from the North Sea. We are aware that work is currently being undertaken in the southern North Sea (Trimmer et al., 2005) and consider that the significance of the effects of trawling on nutrient cycling and localised fluxes must be addressed in North Sea studies (Percival et al., 2005).

Beam trawls, especially large beam trawls with tickler chains or a chain matrix, are amongst the most disruptive gears to benthic habitats and processes (e.g. Collie et al., 2000; de Groot & Lindeboom, 1994).

3.3.1.2 Plankton (phytoplankton and zooplankton)

To the best of our knowledge there are no significant effects of fishing on plankton (phytoplankton or zooplankton). While we acknowledge that change in the population size and distribution of plankton feeding members of the other components may itself be a consequence of fishing effects, there is no known evidence that this is a significant driver in the structuring of North Sea plankton. Changes in the abundance of fish and benthos, from the direct and indirect effects of fishing, will alter the total amount and spatial distribution of larvae produced. In many regions, the seasonal input of meroplanktonic larvae comprises a major part of the zooplankton and this can influence system dynamics through their consumption of phytoplankton and microzooplankton. Similarly, there are certainly occasions when large, gelatinous, plankton are caught in, or macerated by, passage through nets. We are not aware of any studies that allow us to comment on the ecological consequences of this mortality.

3.3.1.3 Benthos

Many of the direct and indirect effects of fishing to benthos are comparable with those of fish communities (Section 3.3.1.4). Benthic invertebrates suffer mortality both in the gears and in the towpath of the gear. Large size, fragile morphology and low mobility have all been associated with increased vulnerability (ICES 2000a, 2002, 2003a). Thus within communities, selective mortality is likely to lead to reduced abundance of large species with low intrinsic rates of increase, and dominance of smaller species with higher intrinsic rates of increase. Changes in size distribution have also recently been described for a number of areas in the North Sea (Jennings *et al*., 2001; Duplisea *et al*., 2002) and the implications of this on secondary productivity have been discussed (Hiddink *et al*., 2006). The interaction between scavenging populations and the increases in moribund material in the towpath of the gear has been described in a number of studies in the Southern North Sea and Irish Sea but the implications of this at the population level and the scale of the North Sea are unknown.

The importance of the physical features of habitats in determining the community structure of benthos is well-documented (Duineveld et al., 1991, Hall et al., 1994). We therefore stress the importance of the overlap between effects of fishing on physical habitat and the effects on the resident benthic communities. The availability of well-defined biotope and habitat maps will significantly improve our ability to assess the effects of fishing on benthos.

Beam trawling produces amongst the most severe impacts on benthos, both because it captures epifaunal and infaunal components but also because of the high mortality associated with contact with this heavy gear (de Groot & Lindeboom, 1994).

3.3.1.4 Commercial fish species and fish community

This summary of the impact of trawling on commercial fish species and fish communities builds on previous work undertaken by WGECO (ICES 2001, 2002), examining the sensitivity of demersal populations to fishing activity.

Within populations, the larger specimens are removed by fishing and over time this selective fishing mortality is expected to lead to changes in growth rate and reductions in age and size at maturity. Within communities, increased mortality leads to reduced abundance of large species with low intrinsic rates of increase (K-selected species), and dominance of smaller species with higher intrinsic rates of increase (r-selected species). Variation in life history characteristics within populations is much lower than among all species in a community, and thus selective effects of fishing on aspects other than abundance are often observed at the community level.

Changes in size distributions in response to exploitation have also been described. As fishing mortality increases on the larger individuals, mean size of individuals in the community drops, and hence small individuals form a larger proportion of the biomass. Consequently, the (negative) slope of size spectra generally becomes steeper while the intercept increases. Size-based approaches such as these provide an effective way of describing gross community responses to fishing, but the structure of the size spectrum and the observed response is based on a combination of factors including: (1) differential vulnerability of larger species; (2) within-population changes in mean size (which in turn implies a reduction in reproductive capacity); (3) genetic changes in life history; and (4) predator-prey relationships within the community. Recent studies have suggested that changes in the size structure of fish communities are as much the consequence of increases in the abundance of small fish, as declines in the abundance of large fish. This suggests that size based indicators are responding to the indirect effects of fishing just as much as to the direct effects (Daan 2005).

In terms of the availability of information there is a considerable difference between the commercial species and the non-target species that together make up the community. For the commercial species, estimates exist of the landings of the fishing fleet and occasionally estimates of by-catch of undersized species are available from discard studies, for the non-target species this information is mostly lacking.

The above describes the generic effects of every type of fishery on the commercial fish species and fish communities but there are differences depending on the métier.

The beam trawl is a heavy gear that uses a series of chains to disturb the sediment surface in order to increase the catch rate of its target species, notably sole and to a lesser extent plaice. The width of the gear is between 4 and 12 m and the height usually no more than half a meter. The mesh-size varies between 80 mm in the southern part of the North Sea to 100 mm in the central or northern part. Most of the effort is concentrated in the SE part of the North Sea. This method of operation and rigging of the gear creates a considerable by-catch of flatfish but also other fish species that occur close to the bottom. To a large extent, the observed changes in the fish community are driven by the removal and mortality of the commercial species.

3.3.1.5 Marine mammals and seabirds

No direct effects of beam trawls on seabirds or marine mammals have been recorded, either in the North Sea or more widely. Some beam trawl fisheries (e.g. for brown shrimp) generate considerable quantities of discarded fish and benthos which is subsequently consumed by seabirds. The implications at the population level vary with area, though it seems likely that seabirds using the southern North Sea for feeding have not been adversely affected, and may have increased in number. Overall population changes need to be interpreted with reference to other factors important in driving variability in these species (ICES, 2003a).

3.3.2 Otter trawling

3.3.2.1 Habitats – physical and chemical attributes & Nutrients

The effects of fishing on habitat are related to the physical disturbance by bottom gears in contact with the seafloor. In summary these include removal of large physical features, reduction in structural biota and a reduction in complexity of habitat structure (leading to increased homogeneity) (ICES 2002, 2003a). The extent of these changes is related to the types of fishing gear being used and the initial level of complexity in both physical and biogenic structure (see Auster & Langton, 1999; Johnson, 2002). See section 3.3.1.1 for more detail.

Traditional otter trawls are not particularly damaging to benthic habitats and processes in sedimentary environments, where the main impact occurs from the otter boards on the seafloor. Generally the impact from otter trawling is considered to be less than that from beam trawling (e.g. Collie et al., 2000; Kaiser et al., in press; de Groot & Lindeboom, 1994). ‘Rock hopper gear’ and any trawl used in a structural complex environment will have more negative impacts and may result in major changes in habitat structure and ecological functioning. Other configurations of the trawl (rollers on the ground gear, tickler chains etc) will all increase the degree of impact on habitat features and benthic processes and may mean that an otter trawl can exert the same degree of impact as a beam trawl.

3.3.2.2 Plankton (phytoplankton and zooplankton)

See section 3.3.1.2

3.3.2.3 Benthos

Benthic invertebrates suffer mortality both in the gears and in the towpath of the gear. Large size, fragile morphology and low mobility have all been associated with increased vulnerability (ICES, 2000a; ICES, 2002; ICES, 2003a). Thus within communities, selective mortality is likely to lead to reduced abundance of large species with low intrinsic rates of increase, and dominance of smaller species with higher intrinsic rates of increase.

Otter trawling is amongst the most impacting gears on epi-benthos, including structural epibiota, as the net and ‘sweeps’ cover a large area and because of the high mortality associated with time spent in the cod end (de Groot & Lindeboom, 1994). Shallow dwelling infauna are also heavily impacted while deeper living forms may be impacted by contact with otter boards or by indirect effects.

3.3.2.4 Commercial fish species and fish community

See section 3.3.1.4 describing the generic effects of bottom trawling on the commercial demersal species and the fish community.

Like the beam trawl, the otter trawl is operated close to the bottom but mostly targets roundfish such as cod, whiting or haddock. Therefore the catch rate of flatfish species is lower than that of the beam trawl while the catch rate of roundfish is higher.

3.3.2.5 Marine mammals and seabirds

Only a few bycatches of seabirds or marine mammals in otter trawls have been recorded, either in the North Sea or more widely. Seabirds and marine mammals have been recorded feeding both within trawl nets and apparently on fish escaping through meshes. Some otter trawl fisheries generate quantities of discarded fish and benthos which is subsequently consumed by seabirds. The implications at the population level vary with area, though it seems likely that seabirds (and possibly some marine mammals) in the North Sea have not been adversely affected, and may have increased in number. Overall population changes need to be interpreted with reference to other factors important in driving variability in these species (ICES, 2003a).

3.3.3 Dredging

3.3.3.1 Habitats – physical and chemical attributes & Nutrients

The effects of fishing on habitat are related to the physical disturbance by bottom gears in contact with the seafloor. In summary these include removal of large physical features, reduction in structural biota and a reduction in complexity of habitat structure (leading to increased homogeneity) (ICES 2002, 2003a). The extent of these changes is related to the types of fishing gear being used and the initial level of complexity in both physical and biogenic structure (see Auster & Langton, 1999 and Johnson, 2002 for reviews). See section 3.3.1.1.

Dredges, especially large multi-dredge rigs, are amongst the most disruptive gears to benthic habitats and processes (e.g. Collie et al., 2000; Kaiser et al., 1997; Thrush et al., 1995).

3.3.3.2 Plankton (phytoplankton and zooplankton)

See section 3.3.1.2.

3.3.3.3 Benthos

Benthic invertebrates suffer mortality both in the gears and in the towpath of the gear. Large size, fragile morphology and low mobility have all been associated with increased vulnerability (ICES 2000a, 2002, 2003a). Thus within communities, selective mortality is likely to lead to reduced abundance of large species with low intrinsic rates of increase, and dominance of smaller species with higher intrinsic rates of increase. The shellfish species targeted by dredges are part of the benthic assemblage and so there is a direct effect on the abundance and size structure of the benthos through their removal.

Dredges are amongst the most impacting gears on benthos, as they are designed to penetrate the seafloor to capture molluscs. They are heavy and so have a high mechanical impact and associated mortality and they have high post-capture damage and mortality in the net (Kaiser et al., 1996, 1997).

3.3.3.4 Commercial fish species and fish community

There is no evidence of concern in relation to by-catch of commercial or non-target fish species in scallop dredges in the North Sea. There is a lack of information on the impact of other dredges and this was felt to be a reflection of the lack of any concerns.

3.3.3.5 Marine mammals and seabirds

No direct effects of dredging on seabirds or marine mammals have been recorded, either in the North Sea or more widely.

3.3.4 Small meshed fisheries

We take small mesh fisheries to be those fisheries employing small mesh to target fish for industrial purposes but NOT fisheries, such as shrimp, which employ a small mesh to target fish for human consumption. Purse seines and light otter trawls are used in the small meshed fisheries.

3.3.4.1 Habitats – physical and chemical attributes & Nutrients

The effects of fishing on habitat are related to the physical disturbance by bottom gears in contact with the seafloor. Typically the gears used in small mesh fisheries do not impact on the seafloor. Purse seines have no direct impact on the sea floor as they are deployed in the water column. Light otter trawls disturb the benthos occasionally but the impact is mitigated as the habitat is generally dynamic sand where the level of natural disturbance is high and the fisheries are seasonal allowing recovery periods. Any indirect effects on the physical and chemical attributes are likely to be small.

3.3.4.2 Plankton (phytoplankton and zooplankton)

See section 3.3.1.2.

3.3.4.3 Benthos

Typically the gears used in small mesh fisheries do not impact on the seafloor and so do not directly impact the benthos, although if one interprets sandeels as being, at least partially benthos, then there is a direct effect via their removal.

3.3.4.4 Commercial fish species and fish community

There has been little evaluation of the consequences of fishing on small mesh targeted species for their main prey. The prey of these pelagic species generally comprises phytoplankton and zooplankton including juvenile fish and eggs (www.fishbase.org; Macer, 1966).

The ICES stomach sampling projects in 1981 and 1991 showed that sandeel, Norway pout and sprat provided more than 50% of the food of saithe and whiting, and between 1-30% of the food of food fish species such as cod, mackerel and haddock (Gislason, 1994). Greenstreet (1996) investigated the diet composition of the main predators in the North Sea to show that industrial fish species form a valuable proportion of the food for predatory fish.

The consumption in the North Sea of sandeels by commercial fish, seabirds and other fish/marine mammals has been estimated as 1.9, 0.2 and 0.3 million tonnes per year, respectively (ICES, 1997a). Cod, haddock, whiting, mackerel, saithe, grey gurnard (*Chelidonichthys gurnardus*) and starry ray (*Raja radiata*) are by far the greatest predators of sandeels (Pope and Macer, 1996; ICES, 1997b). Sandeels comprise 40–60% of the fish biomass consumed and 15–25% of the total biomass in the North Sea (ICES, 1997a). Changes in the size of the sandeel stocks in the North Sea clearly have potential implications for its main predators. However, investigations into the local effect of the closure of an industrial fishery off the east coast of Scotland (ICES, 2004) indicated that that there was no beneficial effect (an increase) on gadoid predator biomass in the region, which was ascribed to the fact that fish predators mainly target 0-group sandeels (Greenstreet, 2006). The fishery targeted older sandeels, so there was a mismatch between the predatory fish needs and the fishery target stock.

No evaluations have been made for the effects of sprat fisheries on the fish community, but at times the sprat fishery has a high by-catch of small herring.

Norway pout can be an important prey item for a number of fish species, but the fishery has reduced in recent years, and fishing mortality is lower than natural mortality and thus it is unlikely that the fishery affects other fish species.

Blue whiting are consumed by a range of piscivores and the species is an important item in the diet of some fish, e.g., cod (Du Buit, 1995). It is difficult to assess the implications on the fish community.

3.3.4.5 Marine mammals and seabirds

The small meshed fisheries for industrial species are typically pursued using purse seines and light otter trawls. Discarding, leading to the provision of material as food subsidies to seabirds and marine mammals, is not an issue since the entire catch is landed. No bycatches of marine mammals have been reported in the small meshed fisheries, but anecdotal evidence exists that a small degree of bycatch occurs (Huse *et al.*, 2003). Seabird bycatch in small meshed gear has been observed in the vicinity of colonies but it is not considered to be a significant form of mortality on populations. (Tasker *et al.*, 2000). The scale of the small meshed fisheries had led to concerns about the impact of the fisheries on seabird and marine mammals populations through indirect mechanisms as they compete for the same resources. Competition is, however, only likely to be an issue in the vicinity of seabird colonies when seabird movement is constrained during the breeding season (ICES, 2003a, 2004).

3.3.5 Fixed gears

We have not considered coastal fisheries in this ToR but offshore bottom set nets, and whelk and crustacean pots are included in this category.

3.3.5.1 Habitats – physical and chemical attributes & Nutrients

The effects of fishing on habitat are related to the physical disturbance by bottom gears in contact with the seafloor. In summary these include removal of large physical features, reduction in structural biota and a reduction in complexity of habitat structure (leading to increased homogeneity) (ICES 2002, 2003a). The extent of these changes is related to the types of fishing gear being used and the initial level of complexity in both physical and biogenic structure (See Auster & Langton, 1999 and Johnson, 2002 for review). See section 3.3.1.1.

Individually the impact from fixed gears on the benthic habitats is small, and caused by individual pots/creels, anchors, weights and ground gear. The largest impacts have been shown to occur when the gear is dragged over the seabed during hauling (Eno *et al.*, 2001). In areas of high habitat structure, particularly biogenic features, the consequences of this can be severe.

3.3.5.2 Plankton (phytoplankton and zooplankton)

See section 3.3.1.2.

3.3.5.3 Benthos

The importance of the physical features of habitats in determining the community structure of benthos is well-documented (Duineveld *et al*., 1991; Hall *et al.*, 1994). In some locations physical impacts from fixed gears, particularly if dragged during hauling, can cause mortality of structural biota and epibenthos. Survival of discarded mobile benthos that have been captured is usually high and so not a major factor.

More so than other gears, many types of fixed gear can 'ghost fish' following loss or jettisoning. A number of studies have quantified these impacts for individual gear items but the scale of the problem remains poorly quantified (e.g. Bullimore *et al.*, 2001).

3.3.5.4 Commercial fish species and fish community

There are no serious concerns in relation to the bycatch of fish in whelk and crustacean pots in North Sea fisheries, although there may be a residual catch in lost pots which continue 'ghost fishing'. Bottom set gill nets are more selective than towed gears and actively target single species, so although bycatch of non-target fish does occur in fixed nets these species are a relatively small proportion of the catch. Such gears are also selective by size and usually do not have high catch rates of juveniles.

3.3.5.5 Marine mammals and seabirds

Fixed gear presents the greatest anthropogenic pressure on marine mammals in the North Sea (see ICES 2005a for a fuller account). Bottom-set gillnets, especially those with a large mesh, are the greatest threat for harbour porpoises. The scale of this bycatch has been non-sustainable at the North Sea population level and may continue to be. Other fixed nets (e.g. salmon drift nets) also catch these marine mammals. Set nets also pose an indirect pressure on seals, as seals are perceived by fishermen to depredate enmeshed fish and are consequently shot be fishermen. In some areas, this shooting is believed to have caused population declines.

It is important to have access to reliable up-to-date information on local population size, distribution and mortality of each species in order to be able to assess the significance of the level of mortality to marine mammals. Considerable evidence has been collected on bycatch, but currently available information on marine mammal population size in the North Sea is about ten years old; the results of a survey carried out in summer 2005 will shortly become available.

Bycatch of large numbers of seabirds has been recorded in fixed nets in the past, especially in the Kattegat. There have been few recent reports on these fisheries and is thought that the scale of bycatch in the North Sea as a whole is much less than in previous years. At a scale of some individual ICES rectangles, this impact may though remain high.

Elsewhere, lines used to mark traps and pots are a significant hazard for some whales, but this does not seem currently to be a problem in the North Sea, probably due to the depleted state of North Sea whale populations. Similarly, bycatch on long-lines (of fishing hooks) is the major cause of decline of some seabird populations. Interactions in the North Sea have not been studied, but this metier is not widely used in the North Sea, so it is unlikely there is a great effect.

3.3.6 Pelagic gears

Purse seines are often used to pursue pelagic fisheries, and when operated inshore they may contact the seafloor. In general, we have considered purse seine fisheries under the pelagic gear type but note that the impacts inshore may also include effects similar to light otter trawls and the information contained in the otter trawl sections should be considered in those cases.

3.3.6.1 Habitats – physical and chemical attributes & Nutrients

The effects of fishing on habitat are related to the physical disturbance by bottom gears in contact with the seafloor. By definition the gears used in pelagic fisheries do not impact on the seafloor and there is no evidence of disruption to pelagic habitat features (e.g. fronts).

3.3.6.2 Plankton (phytoplankton and zooplankton)

See section 3.3.1.2.

3.3.6.3 Benthos

By definition the gears used in pelagic fisheries do not impact on the seafloor and so do not directly impact the benthos. However, if the catch is slipped, i.e. is released after the gear being closed, as may happen in e.g. purse-seine fisheries (ICES, 1991), this may cause considerable local harm to the benthos in terms of organic enrichment and disturbance to the benthic community.

3.3.6.4 Commercial fish species and fish community

The pelagic trawls are only operated in the water-column and have therefore only negligible by-catches of demersal species. Moreover as this type of fishery targets schools of fish it is a relatively “clean” fishery with considerably less by-catch of non-target fish species than the bottom trawls. Inshore use of purse seines in the Skagerrak may take demersal fish as by-catch in shallow areas, especially in inner parts of fjords. When used offshore catches are relatively clean with little by-catch of non-target species (Arrhenius et al. 1998).

Pelagic fish are of course a component of the fish community and considerable changes in the size composition and trophic structure within pelagic fish have been documented. The cause of these changes is less certain, but the fishing down of the larger piscivorous individuals seems likely to have resulted in the observed changes in abundance and size structure (Heath 2005).

3.3.6.5 Marine mammals and seabirds

Bycatch of marine mammals (seals, whales and dolphins) has been recorded in several areas globally, but in relatively small numbers in the North Sea. As most pelagic fisheries have a relatively low rate of discarding, it is not believed that they provide significant food subsidy to scavenging birds or mammals.

3.4 Spatial distribution of fishing effort by gear type

Having considered the generic effects of various fisheries, in order to make a specific assessment of the effects in a particular area requires knowledge of the types of fishing gear being used and the intensity of use. In this section we consider the best available data on the levels of fishing effort (or if this not available landings) for each gear type at the level of the ICES rectangle.

Two EC projects have attempted to assemble “international” effort databases in an attempt to describe the spatial distribution of fishing activity across the North Sea, and so start the processes of estimating spatial variation in fishing impact. The earlier “Biodiversity” study covered the period 1990 to 1995, and provided data for two main gear categories; Otter Trawl and Beam Trawl. The more recent “MAFCONS” project has assimilated data for the period 1997 to 2004, however only for the period 1998 to 2002 are the data complete at present. This latter project aggregated data at four main gear categories; Beam trawl, Otter trawl targeting fish, Otter trawl targeting *Nephrops* and Seine Gear.

The database constructed by the “Biodiversity” project included data supplied by The Netherlands, Germany, Norway, Denmark, England and Scotland. The “MAFCONS” project did not include a Danish partner and their database therefore only included data from the other five countries listed. Both projects attempted to obtain effort data from other non-participating countries, but unsuccessfully in both cases. For both projects the main focus of research was directed towards demersal fish and benthic invertebrate communities, and project consortia included the countries whose fleets had the greatest potential impact on these communities in the North Sea.

3.4.1 Beam trawling

Two beam trawl fishing effort distribution maps are provided, one covering the period 1990 to 1995 based on the “Biodiversity” database (Figure 3.4.1.1) (Jennings et al. 1999) and the second covering the period 1998 to 2002 based on the “MAFCONS” database (Figure 3.4.1.2). Both show average annual hours fishing over the periods involved. The spatial distributions of beam trawl effort in the two periods are almost identical, the only difference being reflected in an overall reduction in total annual beam trawl effort between the early 1990s and the later period. Beam trawling primarily occurs in the southeastern North Sea.

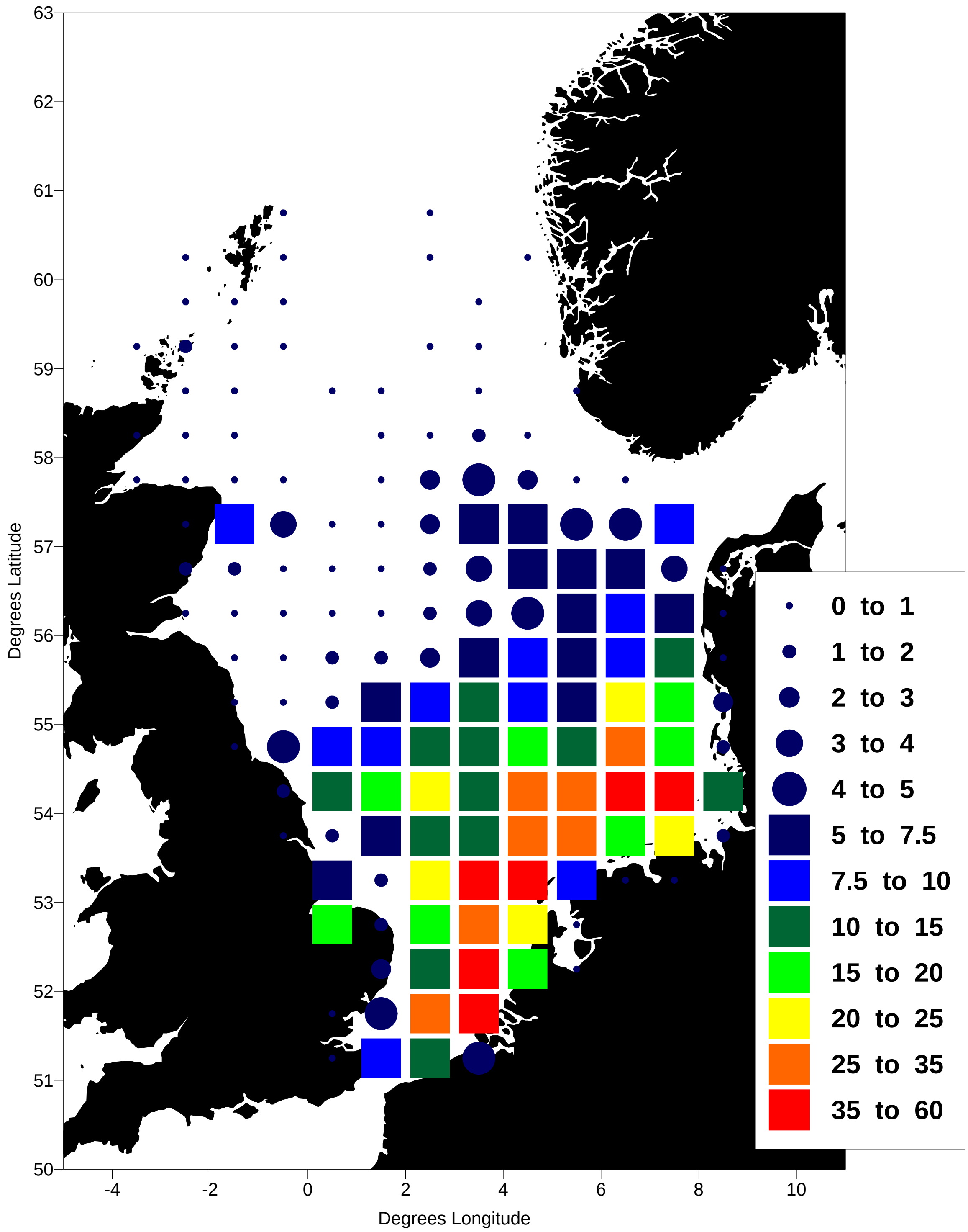


Figure 3.4.1.1. Distribution of average annual (calculated over six-year period 1990 to 1995) Beam trawl effort (1000 hrs.yr^{-1}). Data from Jennings et al. (1999).

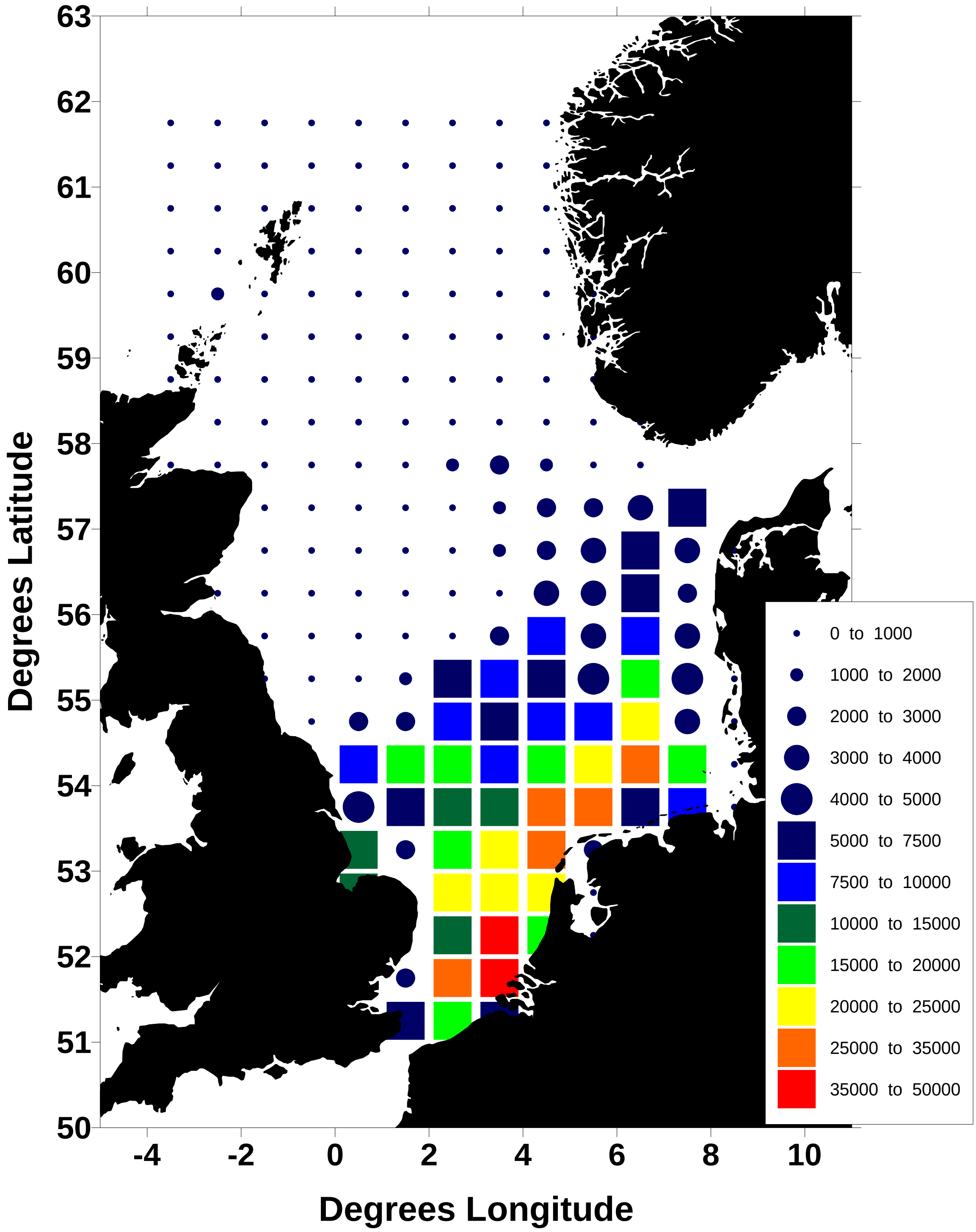


Figure 3.4.1.2. Distribution of average annual (calculated over five-year period 1998 to 2002) Beam trawl effort (hrs.yr^{-1}).

3.4.2 Otter trawling (for both fish and Nephrops, and including Seine Gears)

Figure 3.4.2.1 shows variation in average annual trawl activity derived from the "Biodiversity" database covering the period 1990 to 1995. This map includes all the major otter trawl gears in use at the time and so combines otter trawling directed at fish with otter trawling directed at *Nephrops* (Jennings et al 1999). The later MAFCONS project maintained the distinction between these two quite different types of fishing activity. Thus, for the period 1998 to 2002, Figure 3.4.2.2 shows spatial variation in average annual trawl activity directed towards fish, while Figure 3.4.2.3 shows spatial variation in average annual trawl activity directed towards *Nephrops*. Again little difference in spatial pattern between the two time periods is apparent. Unlike beam trawl, otter trawling principally takes place in the northwestern North Sea.

Spatial variation in the distribution of international seine gear activity has not previously been published. Because of the direct contact of the seine gear coils with the seabed, and fact that the gear relies on the disturbance of the seabed sediment in order to herd fish into the path of the closing seine, this gear in all likelihood has a direct effect on benthic invertebrates within the circle of the gear. The MAFCONS project compiled data for this gear. Since Seine fishing is closest in resemblance to otter trawling, these data are included here (Figure 3.4.2.4).

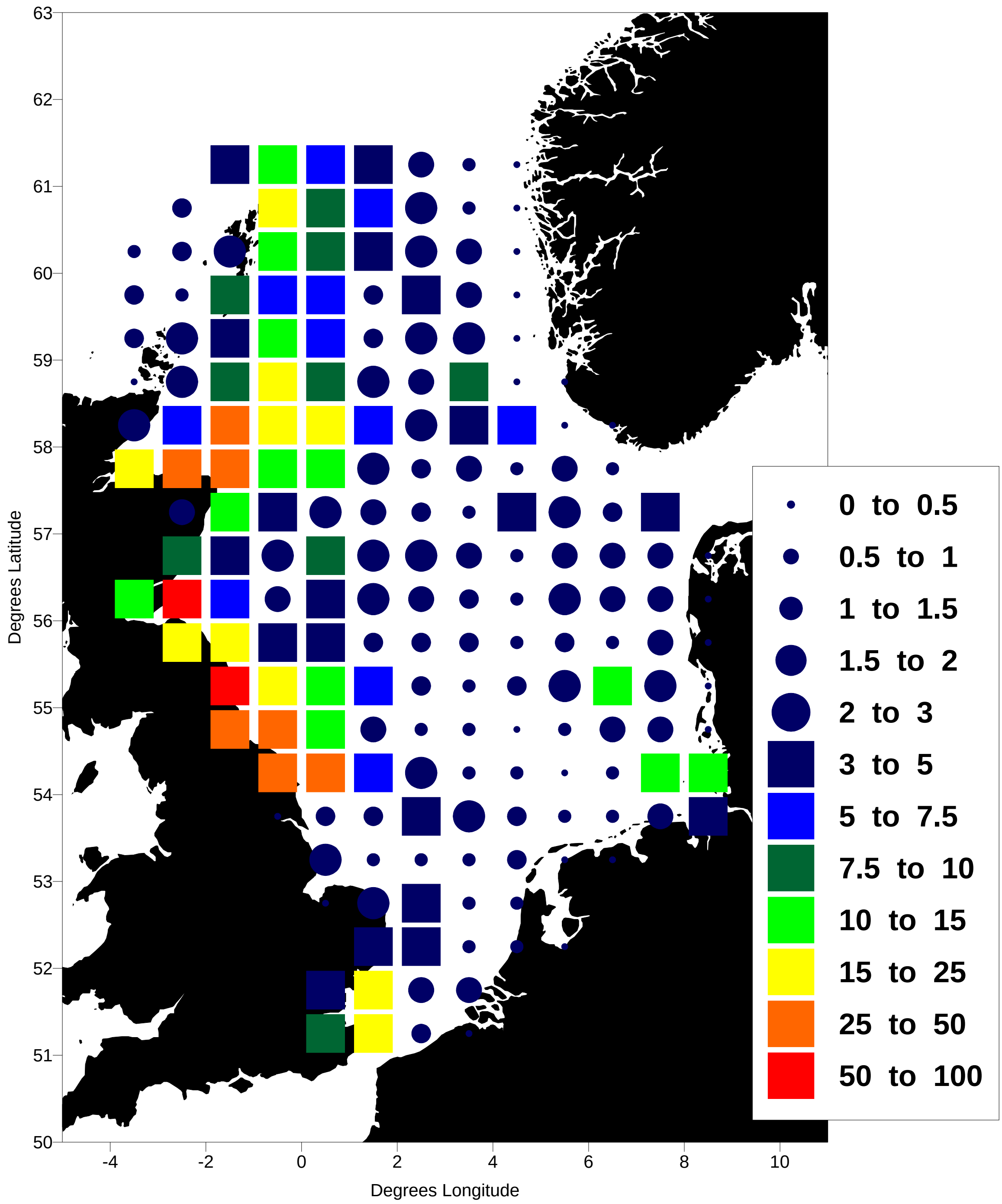


Figure 3.4.2.1. Distribution of average annual (calculated over six-year period 1990 to 1995) Otter trawl effort directed at fish and Nephrops (1000 hrs.yr^{-1}). Data from Jennings et al.(1999).

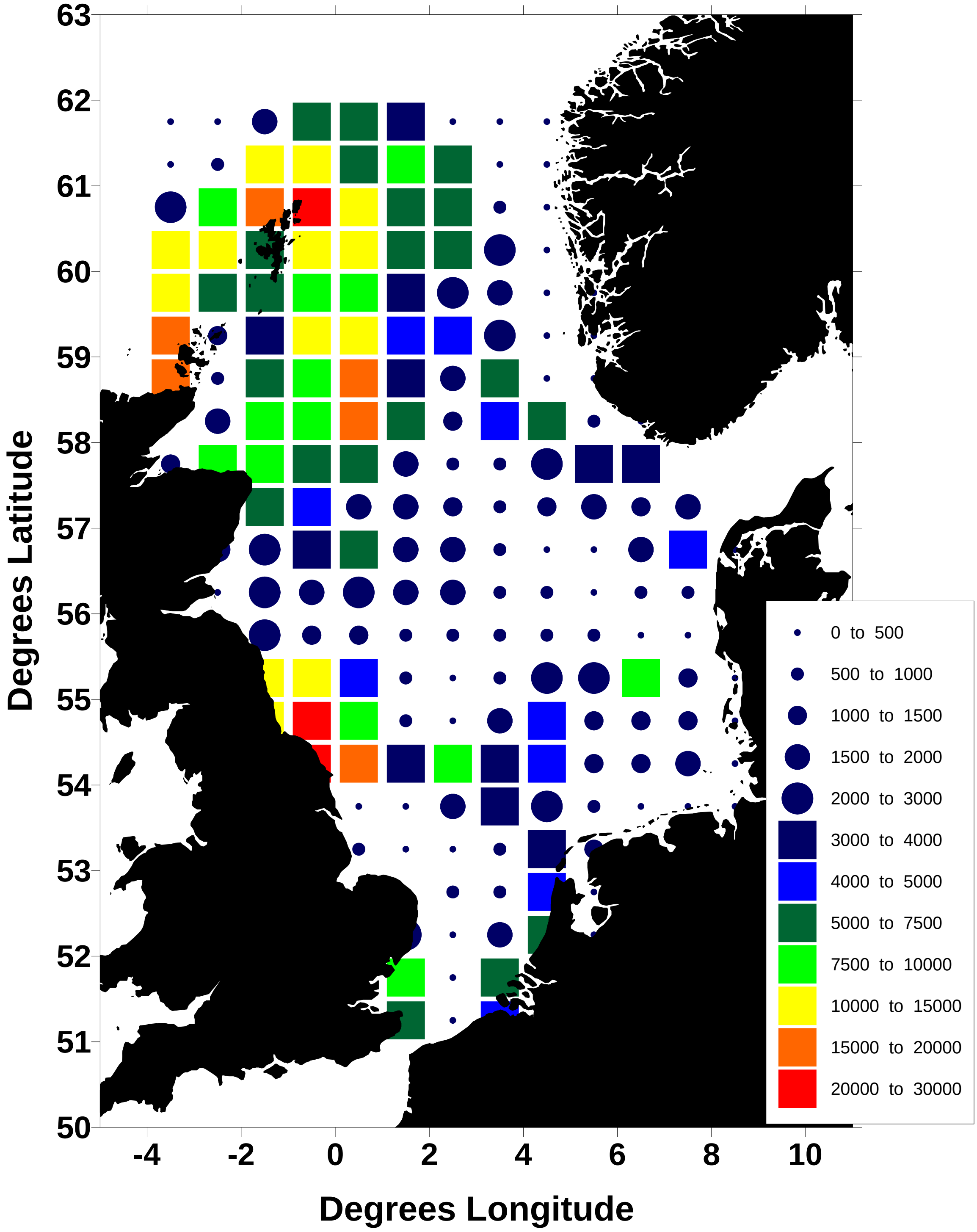


Figure 3.4.2.2. Distribution of average annual (calculated over five-year period 1998 to 2002) Otter trawl effort directed at fish (hrs.yr^{-1}).

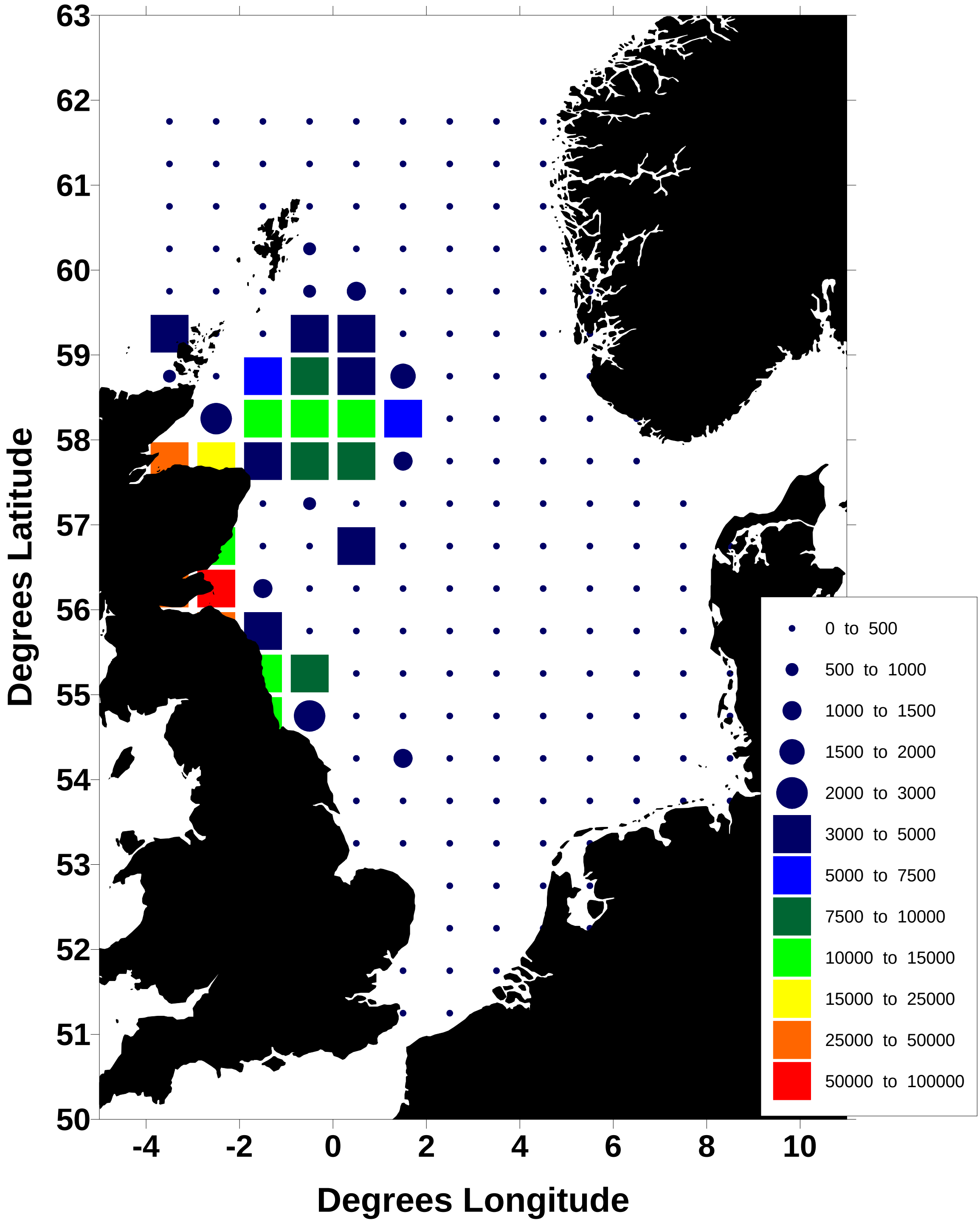


Figure 3.4.2.3. Distribution of average annual (calculated over five-year period 1998 to 2002) Otter trawl effort directed at Nephrops (hrs.yr^{-1}).

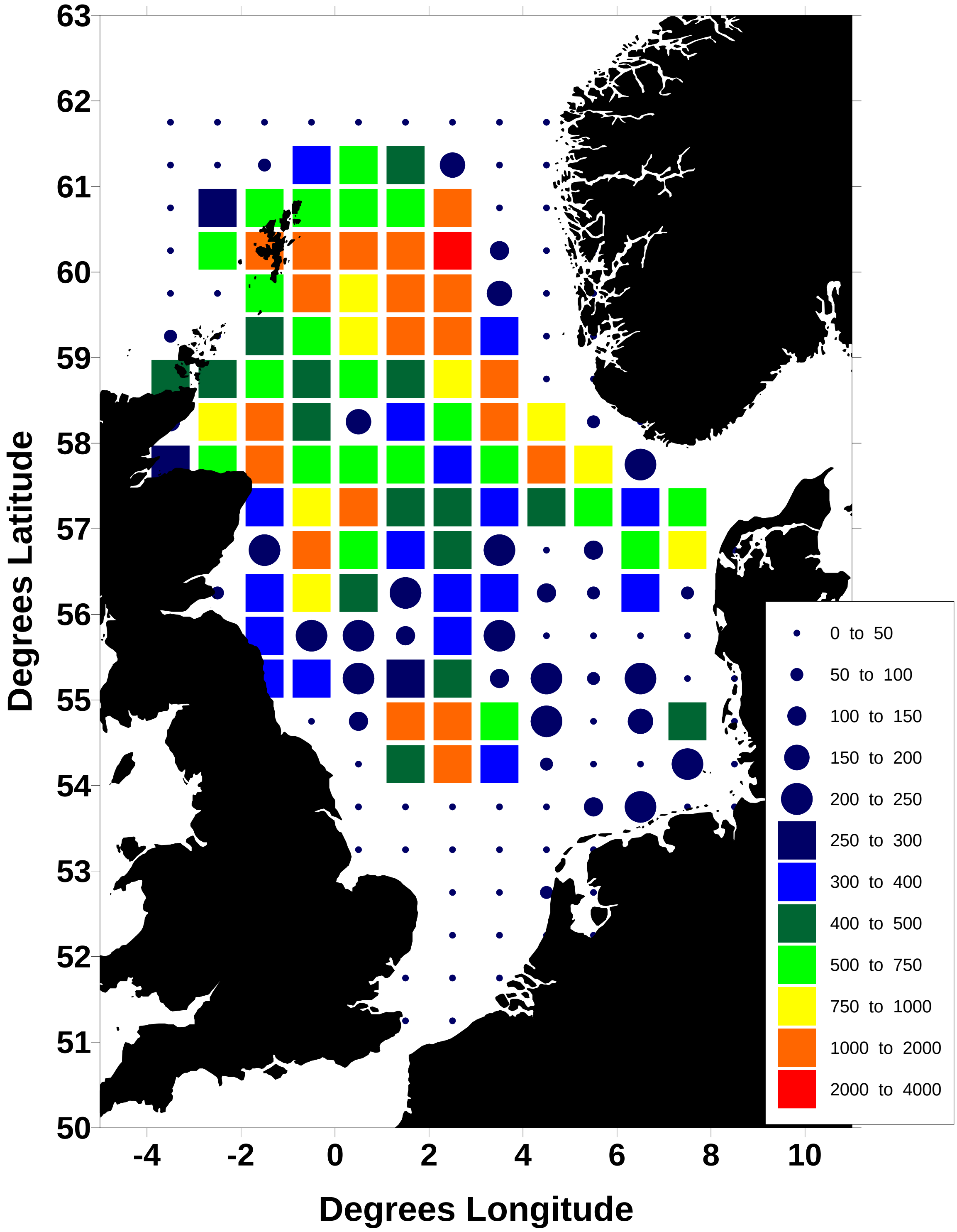


Figure 3.4.2.4. Distribution of average annual (calculated over five-year period 1998 to 2002) Seine gear effort (hrs.yr^{-1}).

3.4.3 Dredging

Fishing activities by dredges for common or blue mussel (*Mytilus edulis*), cockle (*Cerastoderma edule*), and clam species (*Spisula solida, S. subtruncata*), occur in the coastal zones and estuaries of the east coast of England, the French Channel coast, Denmark, and The Netherlands. Scallops, mainly the great scallop (*Pecten maximus*), are fished around Shetland and Orkney, in the Moray Firth (east of Scotland) and off Norway. On the east coast of

England (in ICES Division IVb) recent annual scallop dredge effort by UK (English &Welsh) vessels has been generally less than 100 days fishing per year.

3.4.4 Small meshed fisheries

The distribution of catches by ICES rectangles in the small meshed fisheries in the North Sea is based on logbook data or sales slips.

3.4.4.1 Landings and effort in the North Sea sandeel fishery

The distribution of sandeel catches by year and ICES rectangles, seen in Figure 3.4.4.1.1, is based on logbook data or sales slips from Danish, Norwegian, Scottish and Swedish vessels. These data have been presented in the reports of the ICES Working Group on the Assessment of the Demersal Stocks in the North Sea and Skagerrak, as maps of total international catches of sandeels by year, quarter and ICES rectangle (see e.g. ICES 2006a).

Effort by year and ICES rectangles for the Danish vessels fishing sandeels, seen in Figure 3.4.4.1.2, is based on Danish log book data.

Allocation of effort to ICES rectangles is only possible for the Danish sandeel fleet, as the effort for vessels from other countries only exist as a North Sea scale figure or by the two areas northern and southern North Sea. Further, vessels from different countries fishing sandeels target different areas of the North Sea, i.e. the Norwegian vessels tend to fish more in the northern part of the North Sea than e.g. Danish vessels. The total effort for other countries can therefore not be distributed to ICES rectangles using the Danish effort or landings data.

There is a high degree of correspondence between the distribution of effort and landings in this fishery and it seems likely that this is a characteristic of small mesh fisheries. For the other North Sea small mesh fisheries only landings data are available and so they are used as a proxy for effort.

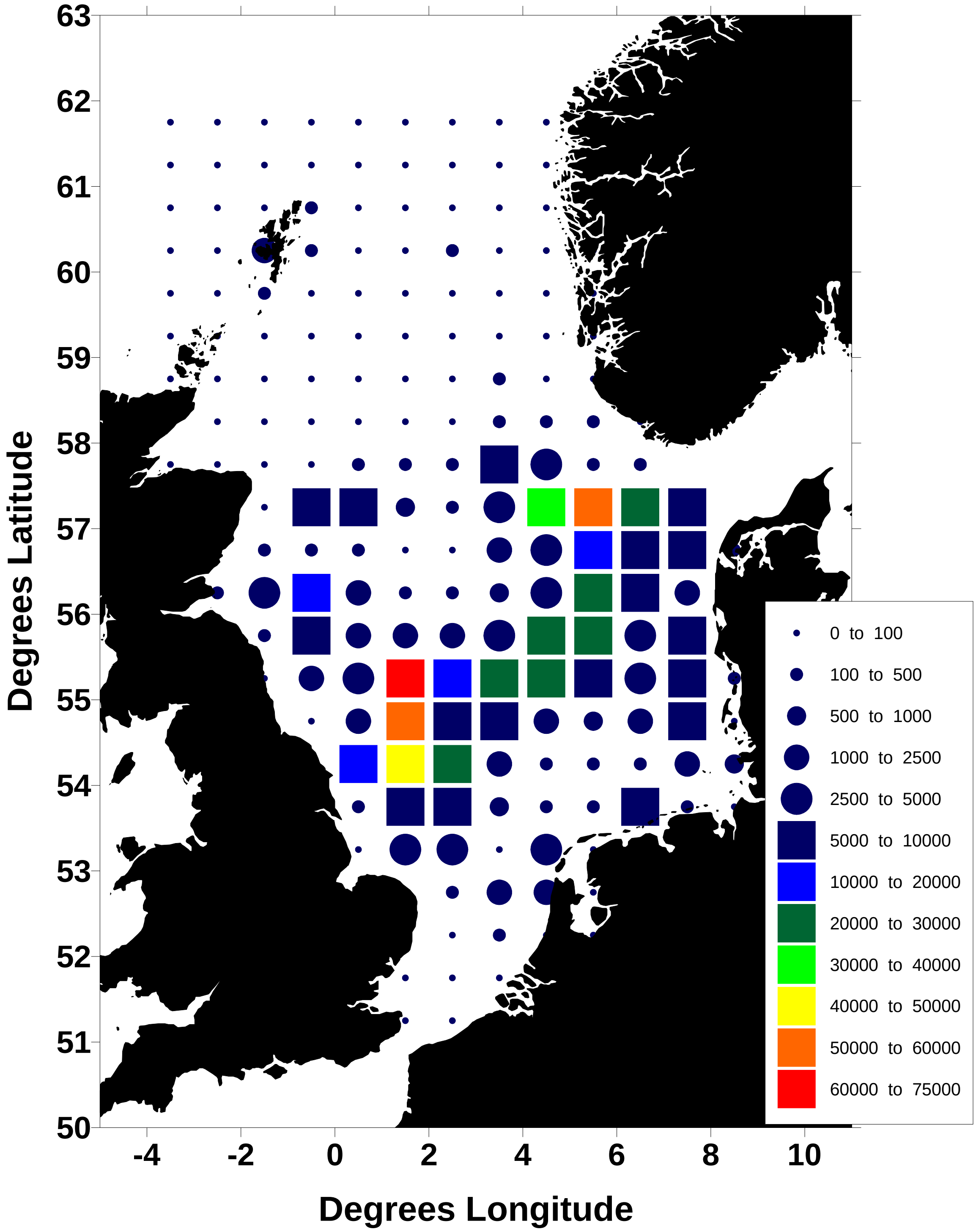


Figure 3.4.4.1.1. Distribution of average annual (calculated over five-year period 1999 to 2003) Sandeel landings (t.yr^{-1}).

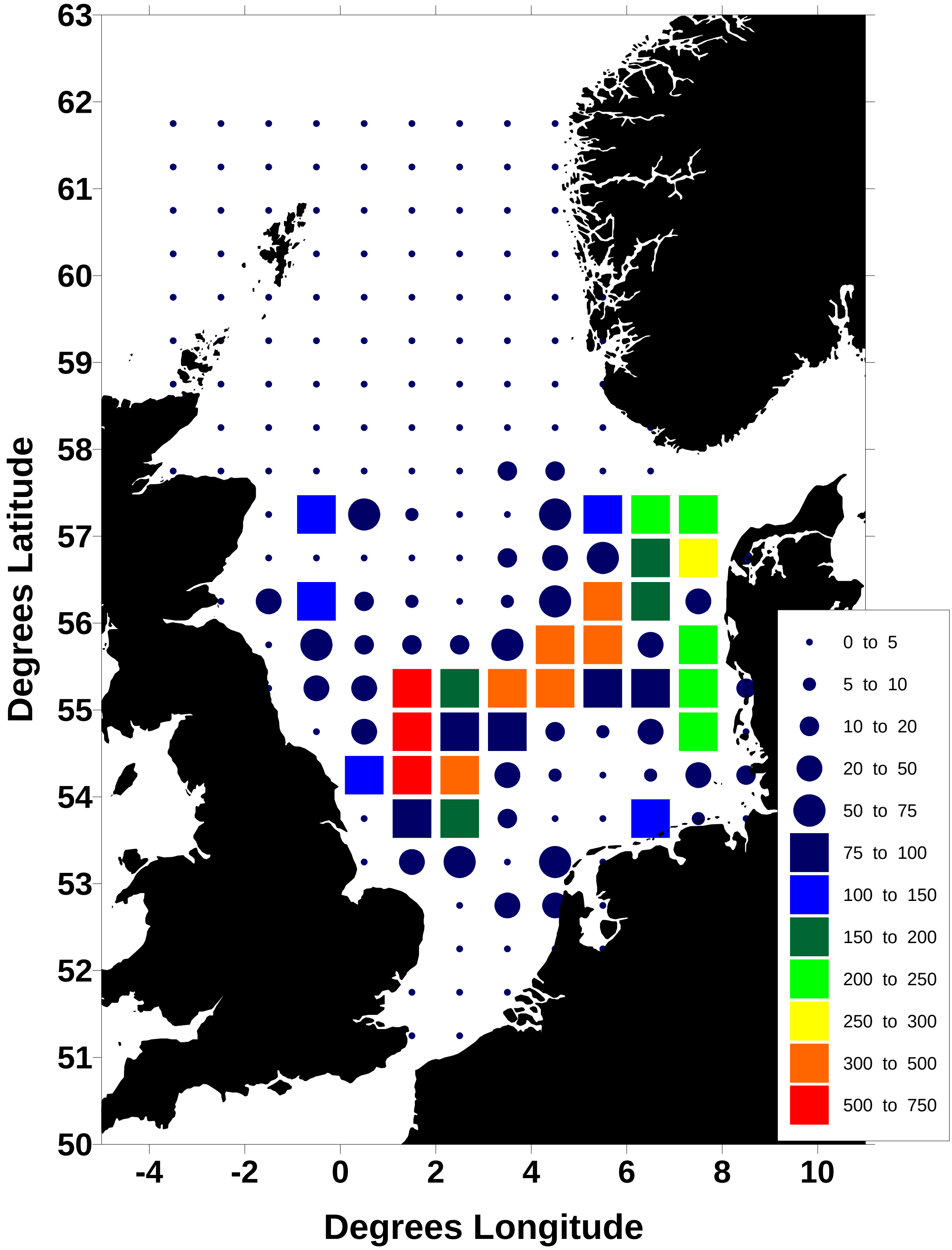


Figure 3.4.4.1.2. Distribution of average annual (calculated over five-year period 1999 to 2003) Danish fishing effort directed at sandeels (d.yr^{-1}).

3.4.4.2 Landings and effort in the North Sea Norway pout fishery

For the Norway pout fishery in the North Sea the WG was only able to get data on international landings of Norway pout by year and ICES rectangle, effort data were not available. The distribution of Norway pout catches by year and ICES rectangles, seen in Figure 3.4.4.2.1, is based on logbook data or sales slips from Danish and Norwegian vessels. These data have been presented in the reports of the ICES Working Group on the Assessment of the Demersal Stocks in the North Sea and Skagerrak, as maps of total international catches of sandeels by year, quarter and ICES rectangle (ICES, 2006a).

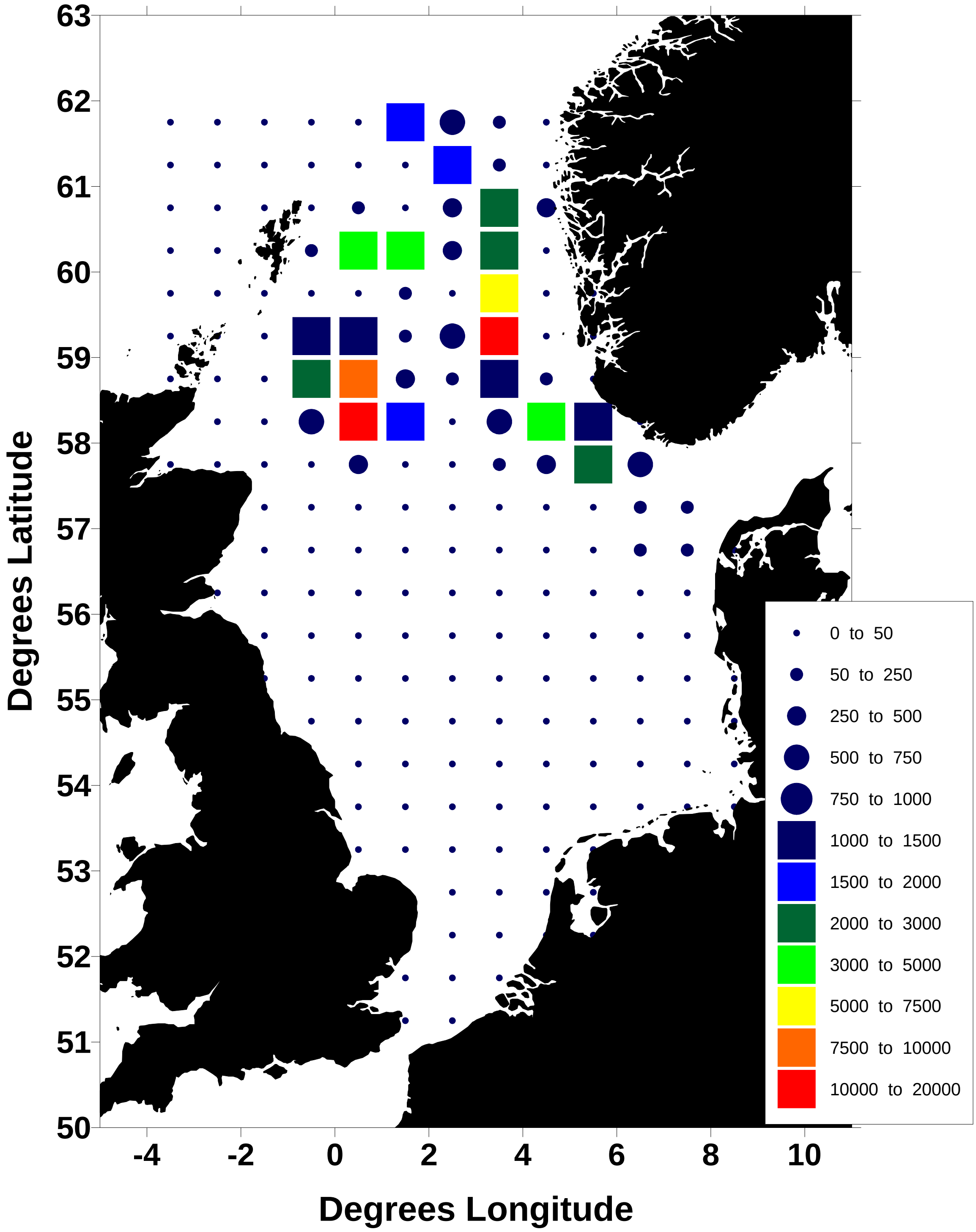


Figure 3.4.4.2.1. Distribution of average annual (calculated over five-year period 1999 to 2003) Norway pout landings ($t.yr^{-1}$).

3.4.4.3 Landings and effort in the North Sea sprat fishery

For the North Sea sprat fishery only landings and no effort data were available. In the North Sea the Danish fleet takes by far the largest proportion of the total international landings of sprat. In the time-period 2000 to 2004 the Danish fleet took more than 97% of total international landings of sprat (ICES, 2005b). Only a proportion of the landings taken by other nations than Denmark could be allocated to ICES rectangles. Therefore only Danish landings data was used for the North Sea sprat fishery by the WG.

Danish landings in the sprat fishery, shown in Figure 3.4.4.3.1, are based on Danish logbook data and sales slips and information about the catch composition in the small meshed fishery. The catch composition is based on samples for species composition taken in port by the Fishery Inspectors. The sprat fishery is here defined as fisheries where at least 50% of the landings in weight consist of sprat. The landings include by-catch of other species than sprat.

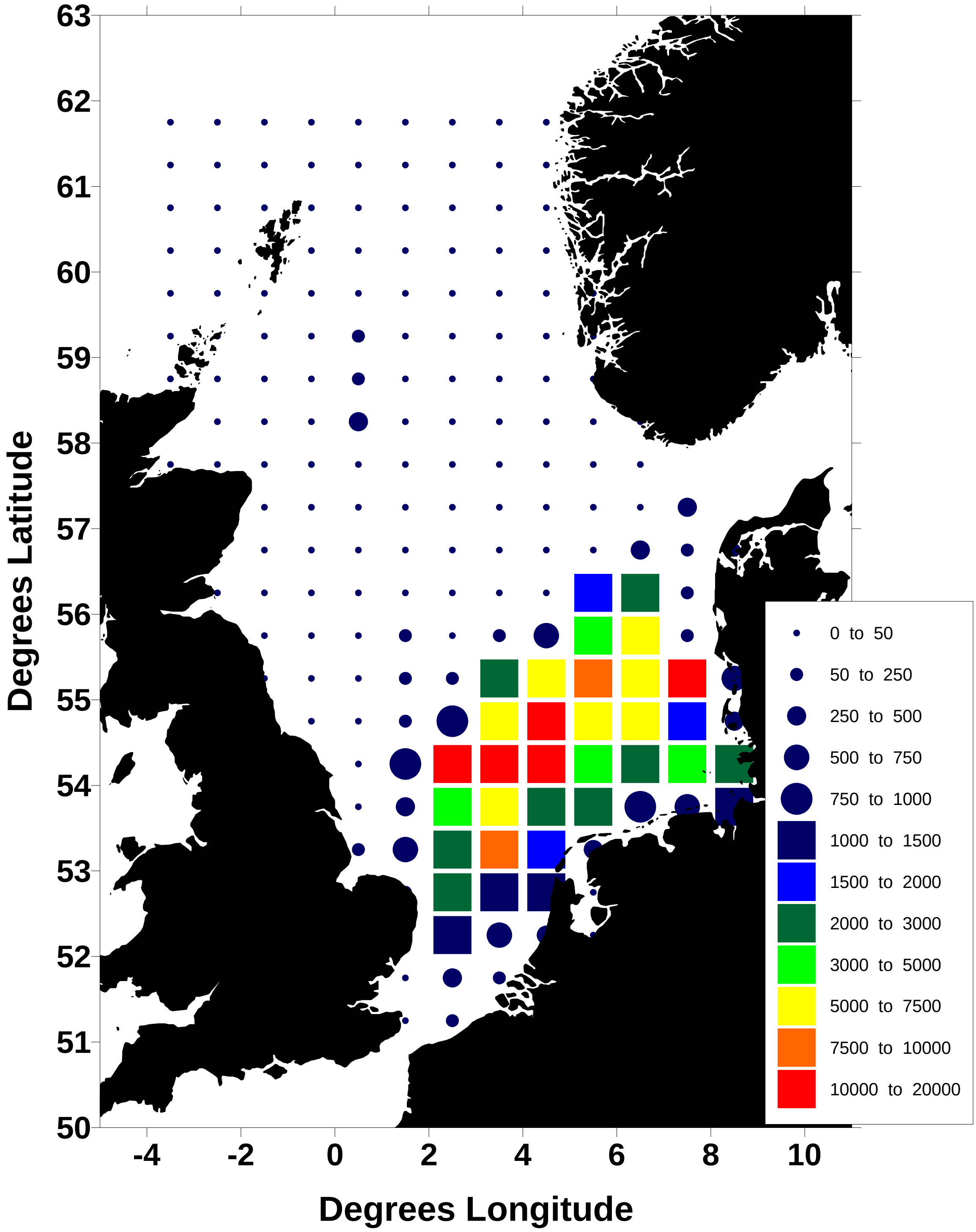


Figure 3.4.4.3.1. Distribution of average annual (calculated over five-year period 1999 to 2003) Sprat landings (t.yr^{-1}).

3.4.4.4 Landings and effort in the North Sea blue whiting fishery

For the blue whiting fishery there are no data available on effort by year and ICES square, but information about catches by ICES rectangle is presented by quarter in the Report of the Northern Pelagic and Blue Whiting Fisheries Working Group (see e.g. ICES 2005c). WGECO was only able to get information about landings by ICES rectangle for 2004 for Denmark and Norway. Danish and Norwegian landings of blue whiting represent 98% of total landings of blue whiting in the North Sea in 2004 (ICES 2005c).

The principle blue whiting fishery occurs in the deeper shelf slope waters to the west of Scotland, Ireland, around the Faroes and towards Iceland. In the North Sea, nearly the entire fishery occurs in the Norwegian trench to the south of Norway (Figure 3.4.4.4.1).

Although the majority of the blue whiting fishery is for adult or full-grown fish, a substantial tonnage of smaller fish is landed in other fisheries. These other fisheries are the Norway pout fishery, a mixed small meshed fishery, and a human consumption herring fishery in the Skagerrak.

Because there are some differences between the 2004 fisheries in which blue whiting are caught, compared to previous year's, additional information about the Danish and Norwegian fisheries in 2004 is given below, based on information from ICES (2005c).

The Danish directed fishery blue whiting fishery is mainly conducted by trawlers using a minimum mesh size of 40mm. Blue whiting is also taken as by-catch using trawl with mesh sizes between 16 and 36 mm for Norway pout, however in 2004 this fishery was very limited. The main Norwegian fishery is a directed pelagic trawl fishery, carried out on and west of the spawning areas west of the British Isles and in the Norwegian Sea using pelagic trawls with minimum mesh size of 35 mm. Blue whiting is also fished in the North Sea and in the southern Norwegian Sea (areas east of 4ºW) in the mixed industrial fishery targeting blue whiting and Norway pout. Before 2004 these vessels were only allowed to fish blue whiting south of 64ºN. These vessels use small-meshed trawls operated close to the bottom (minimum mesh size 16 mm) or pelagic trawls with minimum mesh size of 35 mm.

In 2004, as usual, there was a seasonal progression of the Norwegian blue whiting fishery from the international waters off Porcupine Bank and Rockall in the beginning of the season (January-March) towards the shelf edge in EU zone and the banks in the Faroese waters in the end of the spawning season. The fishery in EEZ of EU was stopped on 26 April 2004 and that in the Faroese zone on 16 June after the quotas in the respective zones were taken. The fishery in the Norwegian EEZ, Jan Mayen zone and international waters was stopped for the period 29 April-23 May; this was at least partly a reaction to large proportions of juvenile blue whiting in the catches in the southern Norwegian Sea. For the rest of the season, no regulation took place.

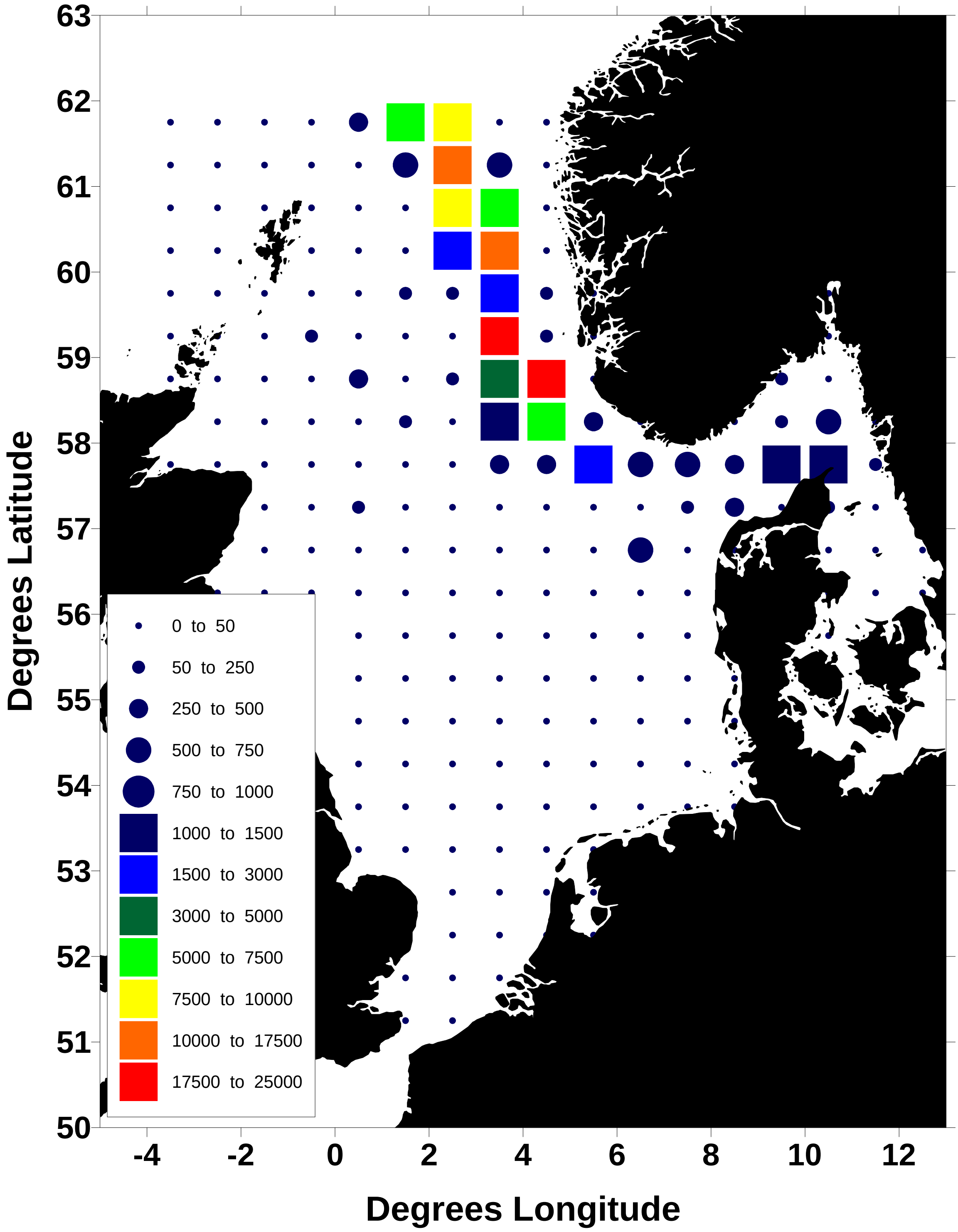


Figure 3.4.4.4.1. The distribution of Danish and Norwegian Blue whiting 2004 landings (tonnes), for details see text.

3.4.5 Fixed gears

Detailed information on effort in fixed gear fisheries in the North Sea is rare or non-existent. In relation to ecosystem effects, ideal data per ICES rectangle (or finer scale) might be a) km.hours of soak time for nets, b) trap.days for pots or other fixed traps (e.g. pound nets), c) number of hooks shot for long-lines.

In relation to these métiers, effort in the net fishery is available (but not accessible by WGECO in time for this meeting) for some, but not all, countries around the North Sea in terms of days at sea by larger vessels undertaking the fishery, or trips (often one or two days in duration) for smaller vessels. This effort can be related to relatively few coastal ICES rectangles for day trips, or to broad sections of the North Sea (e.g. eastern central) for larger vessels. Landings data may be available from this fishery, but relating this to effort will be difficult. Information in relation to pot fisheries probably relates to the number of vessels in the fishery. Long-lines are not in common use in the North Sea, with the possible exception of deeper water fisheries in the north-west. Information on numbers of hooks used is not known.

3.4.6 Pelagic gears

Landings of three main fish species caught by pelagic gear (pelagic trawls and pelagic seines) are presented in Fig. 3.4.6.1. Data represent annual averages of the sum of landings by the countries Denmark, UK (Scotland and England), France, Germany, Ireland, Netherlands and Sweden. Herring landings were primarily from the western and northern North Sea, with local maxima off the UK coast (up to > 10,000 t/yr per ICES rectangle) and an extended region of high landings around the Orkney and Shetland Islands. The northern part of this region also contributed the majority of mackerel landings, which were very low everywhere but in the very northern North Sea. Horse mackerel landings were comparatively low, with a maximum annual average of below 1000 t per ICES rectangle, located in the central-northern North Sea.

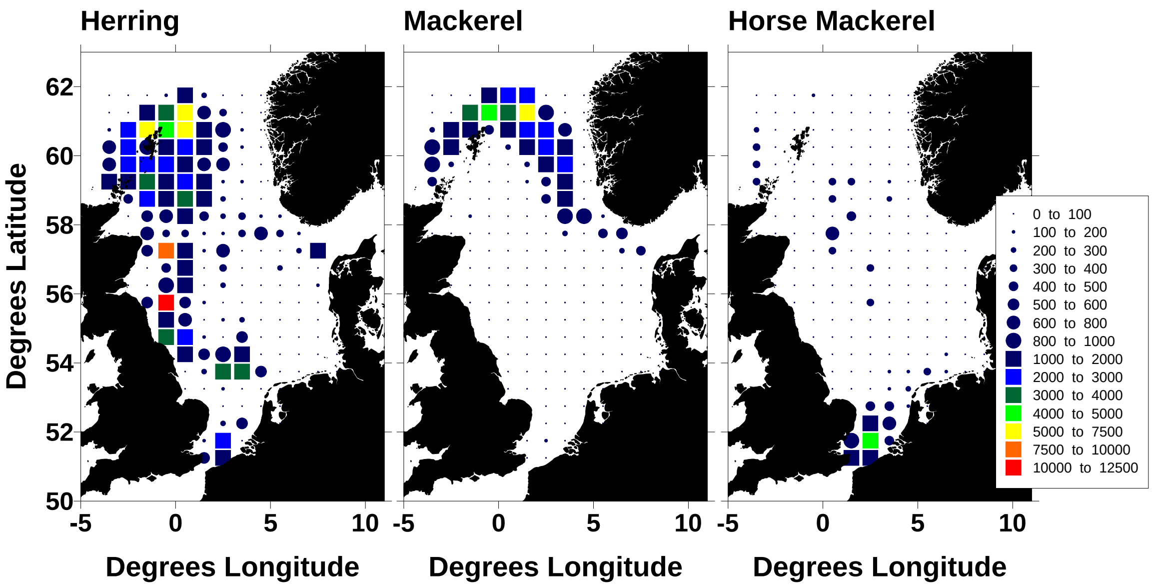


Figure 3.4.6.1. The distribution of herring, mackerel and horse mackerel landings (tonnes) averaged over the years 2000 to 2004. (This analysis has been conducted with the database collated and provided by the STECF subgroup STGST- Cod Recovery Plan; STECF, 2005).

3.5 Direct effects of fishing on the North Sea ecosystem components

3.5.1 Habitats – physical and chemical attributes

Mobile fishing gears can alter the physical structure of the habitat through the input of energy and the movement of particles, including very large particles. The scale of these changes varies depending on the nature of the habitat and the type of fishing operation. In some cases a single fishing operation may permanently alter the habitat.

Direct habitat effects of fishing in the North Sea are only likely to occur in areas where fragile habitat features are exposed to fishing by high impact gears. For the most part dredging, beam trawling and otter trawling occur in habitats which are not sensitive to the physical effects of fishing. Given our incomplete knowledge of seabed habitats this must be interpreted with caution as habitat features such as isolated boulders may occur and be vulnerable in areas of otherwise robust habitat. Topography and occurrence of habitats is more variable in the Skagerrak and Kattegat region in comparison to the North Sea and bottom fishing may impact on more complex habitats in this area.

Biological habitat features are generally highly vulnerable to bottom fisheries and these are considered below (Section 3.5.4) as part of the benthos.

The only way to effectively mitigate the effects of towed bottom gears on sea floor habitats is to spatially separate them by excluding damaging gears from habitat areas. Such a management plan would probably involve exclusion of gears from ALL areas where rare or particularly valuable habitats occur and exclusion from a proportion of other areas in order to ensure the existence of some areas in near natural conditions.

3.5.2 Nutrients

Bottom contacting fishing gears can move sediments and disrupt geochemical processes at the sea floor. In particular, physical turnover of the sediment and alterations in the oxygen environment in the sediments can lead to altered rates of remineralisation of organic matter and nutrient regeneration, and altered rates of nutrient efflux.

In the North Sea, there are good data to suggest that bottom disturbance by fishing gears in muddy sediments and at very high rates can cause significant increases in nutrient efflux (Percival et al., 2005). However, significant effects are likely to be limited to very small areas where muddy sediments and high intensity fishing occurs. In more typical situations field data show that the fishing induced changes are small and insignificant functionally (Trimmer et al., 2005). However, in the Skagerrak and Kattegat fishing occur frequently on muddy sediments and fishing contributes significantly to the resuspension of muddy sediments (Floderus & Pihl, 1990). Eutrophication is considered a severe problem in particular in the Katteget and fishing induced nutrient efflux might well be a contributing factor.

At the North Sea scale, there seems to be no compelling case for management intervention based on the impacts of bottom fishing gears on nutrient dynamics.

3.5.3 Plankton (phytoplankton and zooplankton)

No evidence of North Sea fisheries impacting on North Sea plankton except in that numbers of meroplanktonic fish larvae will have declined as the major stocks have declined.

3.5.4 Benthos

Fishing impacts benthos through direct mortality, injury (leading to disease, lowered fitness), altered food supply, altered predation pressure, and changes in the physical environment (habitat).

In the North Sea comparisons of historic and contemporary data (Robinson & Frid, 2005; Rumohr & Kujawski, 2000), time series (Bremner et al., 2005; Frid et al., 1999a), comparisons of areas differing in their fishing history (de Groot & Lindeboom, 1994;, Dinmore et al., 2003) all suggest fishing induced changes in the benthos. These changes include loss of large fragile forms but also alterations in the relative abundance of a large number of species (Clark & Frid, 2001; Frid & Clark, 2000). Recent advances in our knowledge of the spatial extent of fishing (Section 3.4), the increasing availability of seafloor habitat maps and novel modelling approaches (Section 3.10) will allow the development of management regimes that include explicit consideration of the spatial distribution of benthic impacts. To get a first impression of the actual “footprint” of fishing on benthic invertebrate communities across the North Sea the benthic impact model (Section 3.10) was run using actual mortalities caused by each gear type (Figure 3.5.4.1). “Per fishing event” mortality rates for each of the four main fishing gear categories were derived from Tulp et al. (2005). The first run used gear average mortalities calculated across 12 benthic invertebrate phyla. These mortalities were 0.25 for beam trawl, 0.1 for the two otter trawls and 0.05 for Seine gears. From the model output, it is clear that benthic communities are affected by fishing activity across almost the entire North Sea, but impact is greatest in the southeastern North Sea, where the most damaging of the fishing activities, beam trawling, is most prevalent.

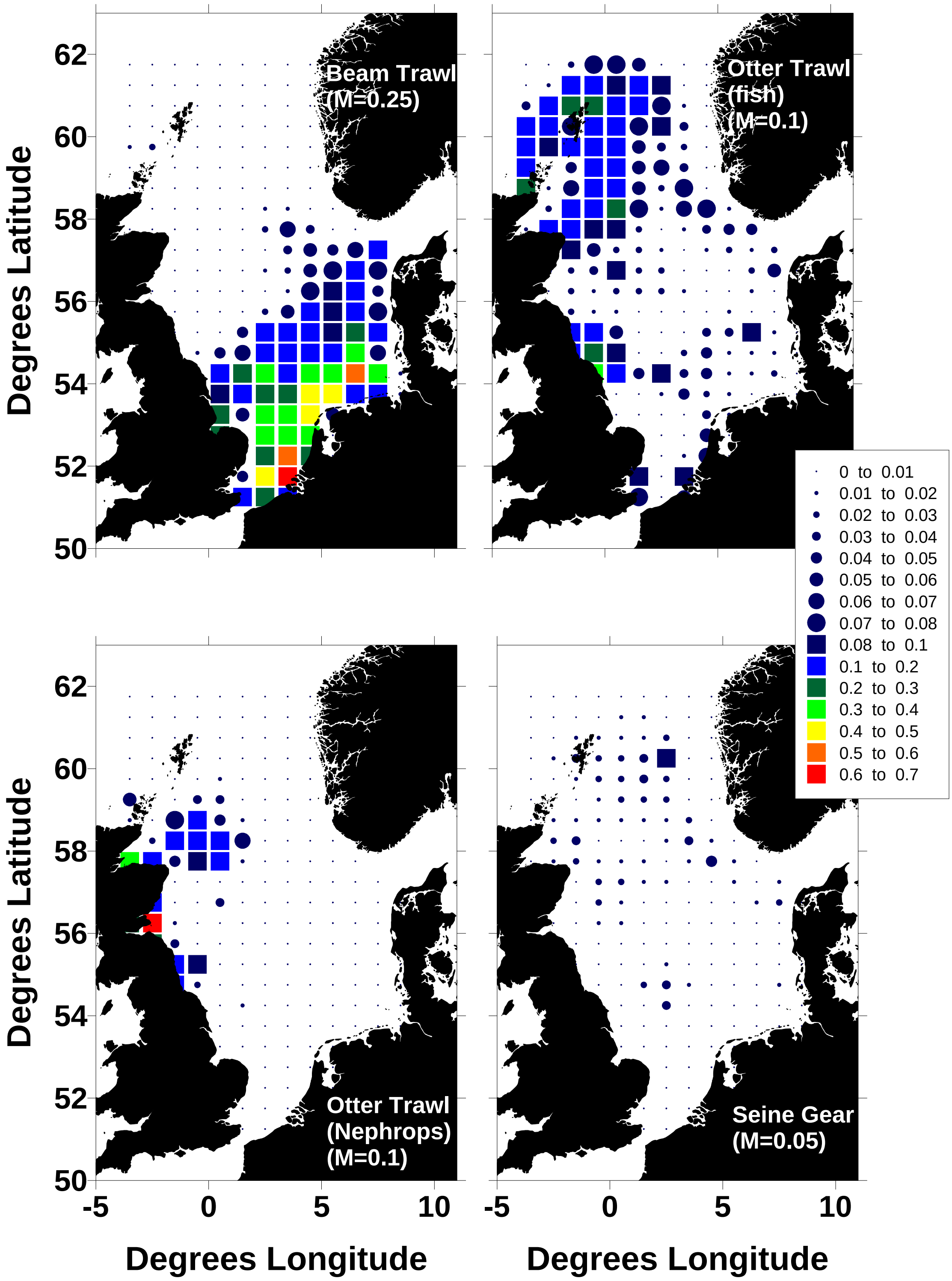


Figure 3.5.4.1. Modelled impact of four major demersal fishing categories on the benthic community of the North Sea. Maps show total modelled annual mortality given the distribution of beam trawl, otter trawl targeting fish, otter trawl targeting Nephrops and Seine gear fishing activity 1998-2002 and "per fishing event" mortality rates of 25%, 10%, 10%, and 5% respectively for each gear type.

The only way to effectively mitigate the effects of towed bottom gears on benthos is to spatially separate them by excluding damaging gears from areas where the benthos are of concern. Such a management plan would probably involve exclusion of gears from ALL areas where rare or particularly valuable benthos occur and exclusion from a proportion of other areas in order to ensure the existence of some areas of near natural dynamics. It would seem beneficial to plan such measures with regard to the delivery of habitat protection (see Section 3.5.1.1), as protection of habitat is likely to deliver a large amount of protection to the associated benthos (Frid & Hall, 2001).

3.5.5 Commercial fish species and fish community

For commercial fish species the direct effects of fishing are described by the catches of the respective species made up of landings and discards. The landings data are usually readily available from the ICES assessment working groups that deal with these species, but for most species this does not apply to discard data. For this analysis, two sources of landings data were available, one collated by the STECF Sub-group SGRST on Evaluation of the Cod Recovery Plan Ref. (STECF, 2005), the other by the MAFCONS project. Although these sources differed somewhat, e.g. in the countries included and the gear types distinguished, they show overall the same spatial distribution of landings. As only the MAFCONS project had spatially disaggregated effort data, the results in this section are based on this data source. Figure 3.5.5.1 shows the spatial distribution in the North Sea of landings and proportion of the catch discarded for cod, haddock, plaice and whiting. These estimates are based on two métiers of the North Sea fishing fleet, i.e. beam trawl and otter trawl. The rectangles for which information on the proportion of the catch discarded is missing are rectangles in which this species was not caught by the survey.

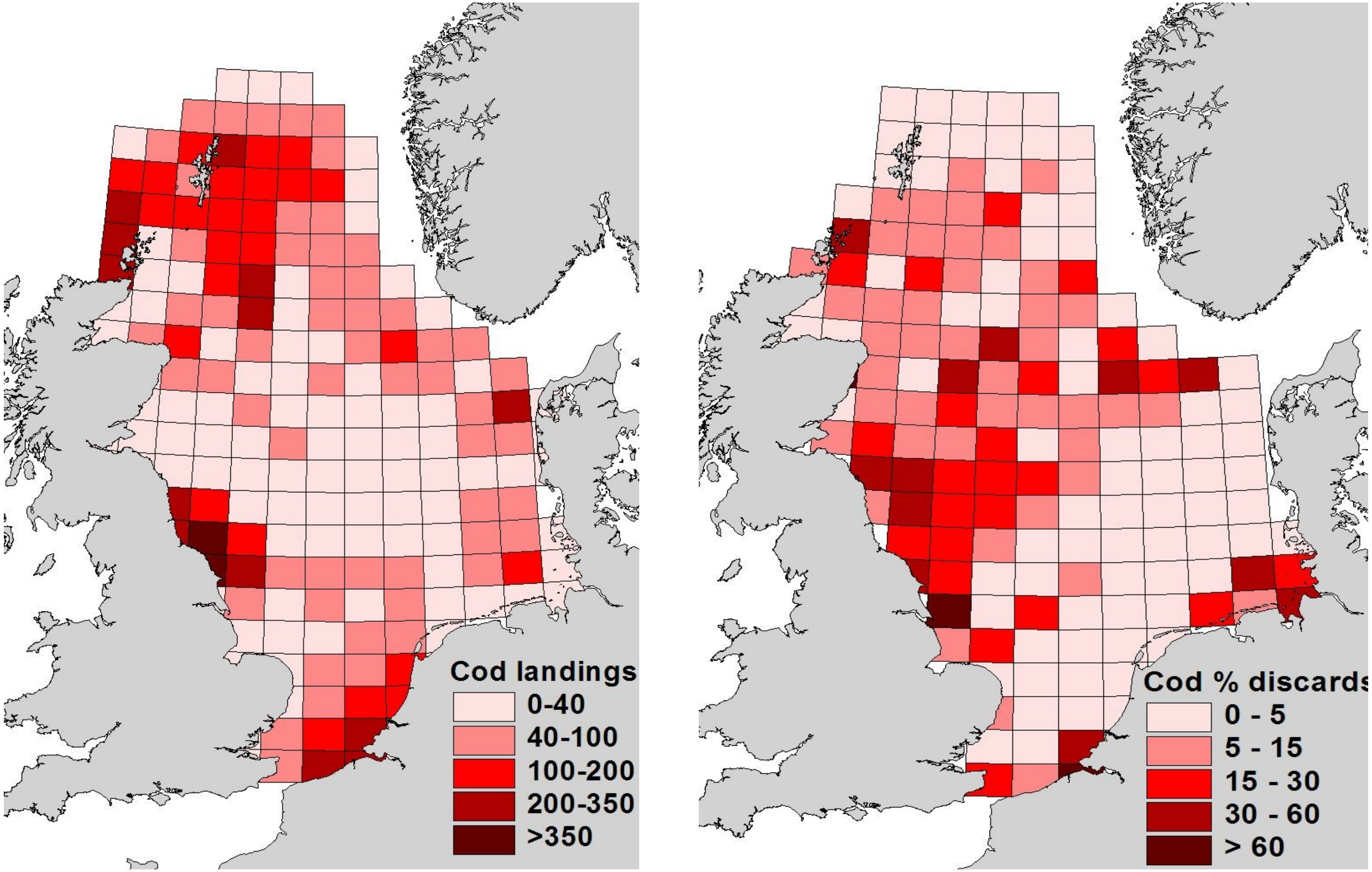
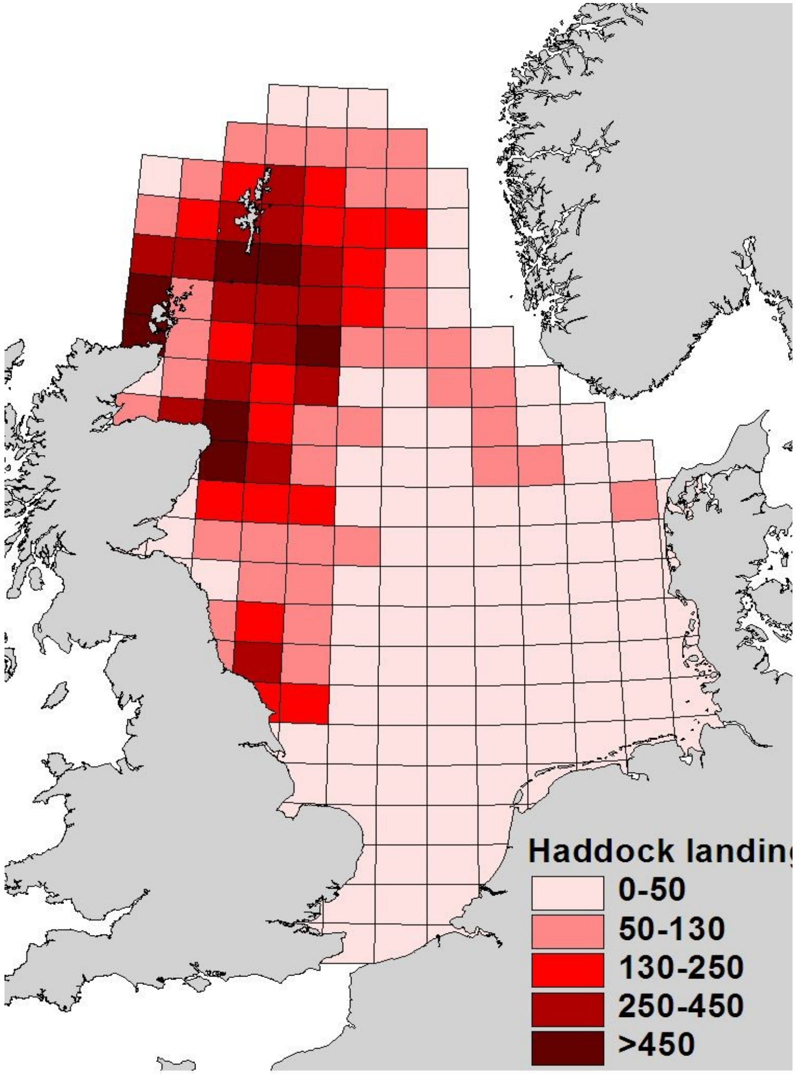


Figure 3.5.5.1. Landings (tonnes) from MAFCONS database and the derived (modelled) discards as percentage of the catch (period 1998-2002).



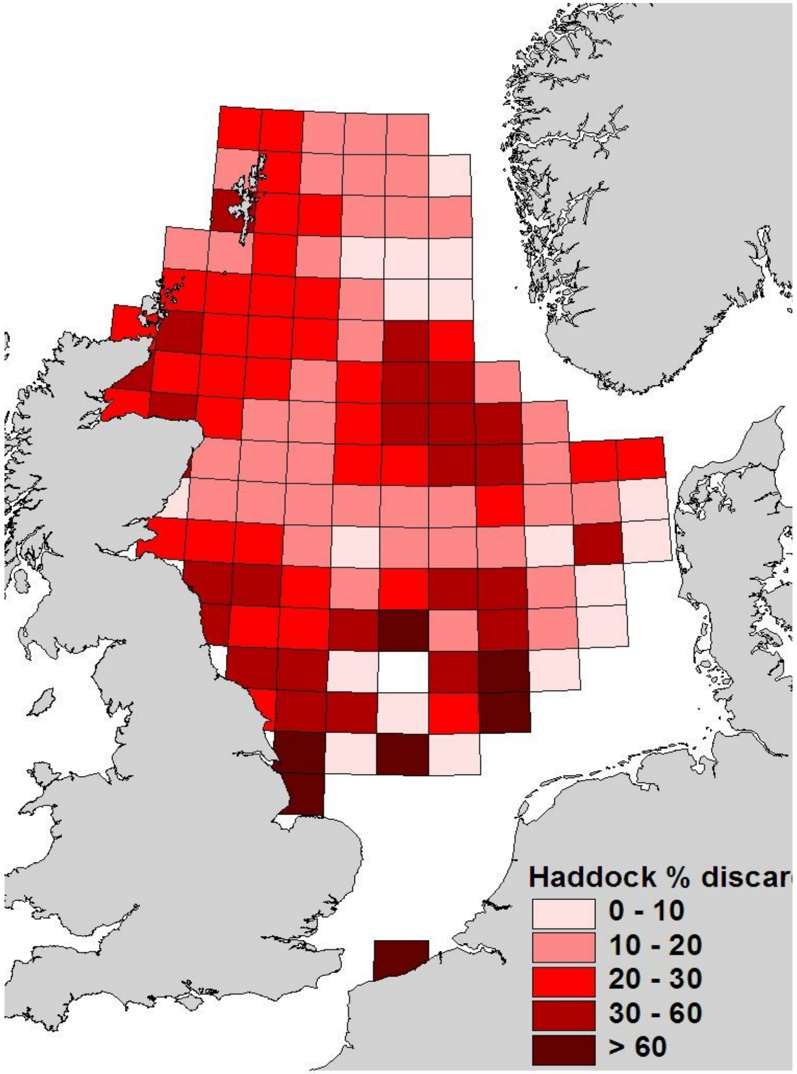
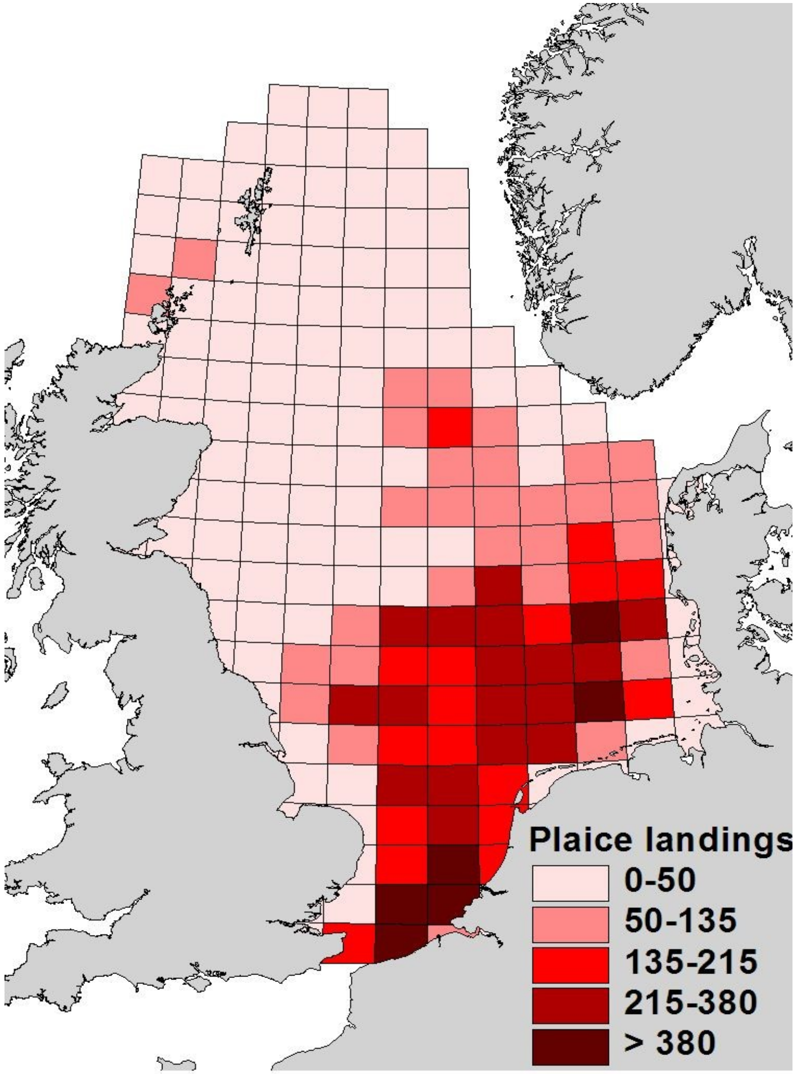


Figure 3.5.5.1. continued



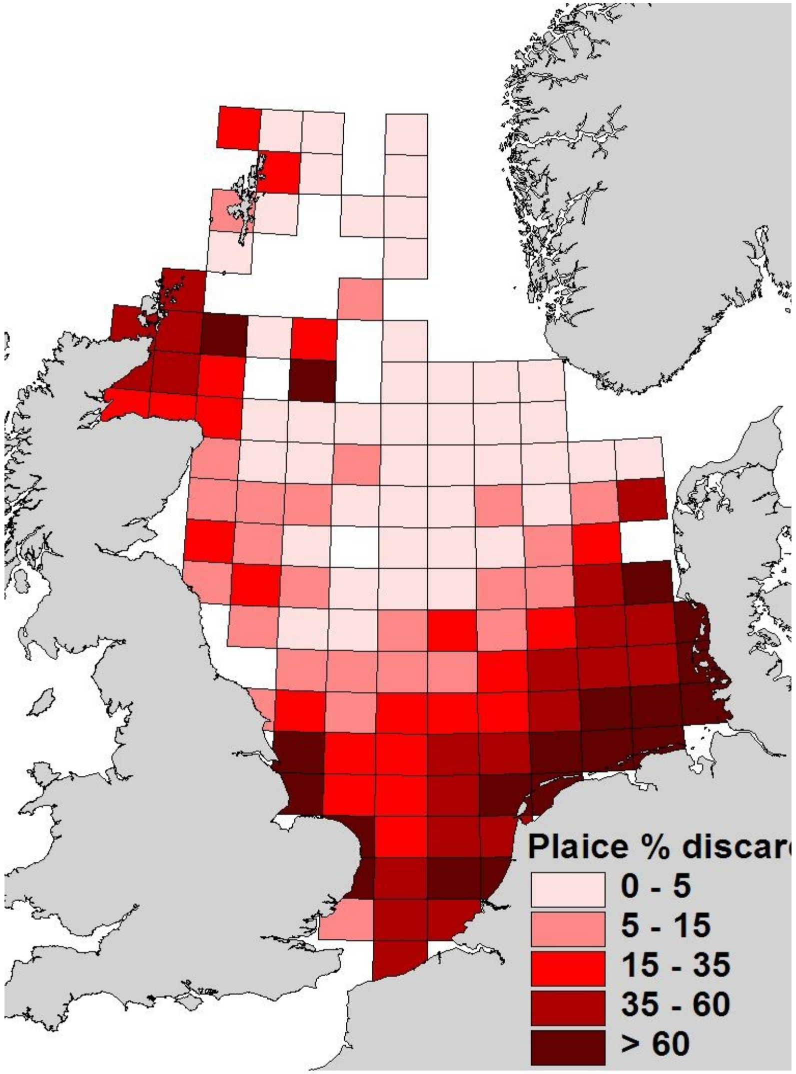


Figure 3.5.5.1. continued.

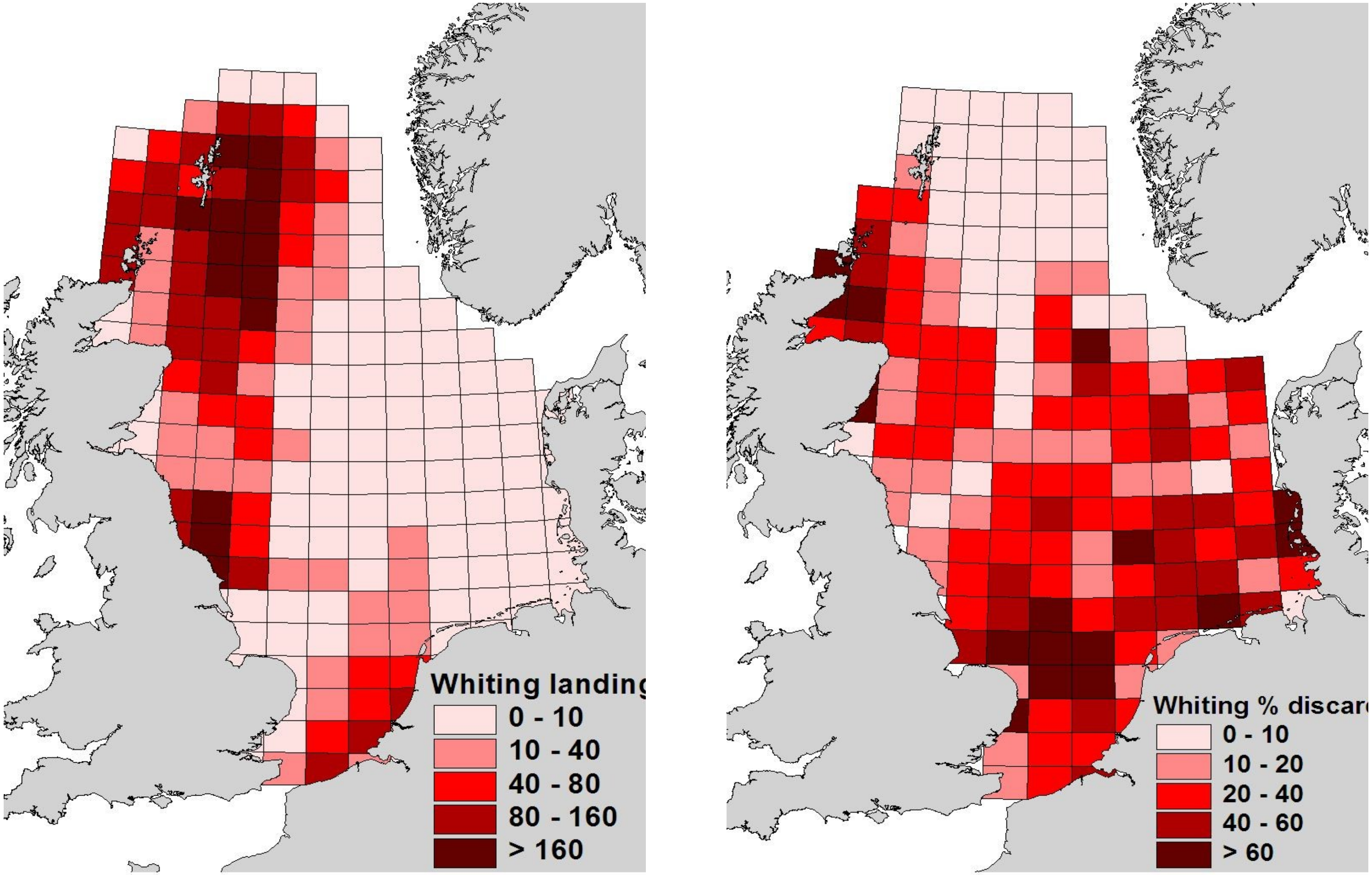


Figure 3.5.5.1. continued.

The impact of fishing on the commercial species is not only determined by the landings but also by the discards. Notably for the latter there is often very little information available and representation is often an issue. For the remainder of the fish community, i.e. the non-target species, fishing impact is almost entirely determined by fish caught and discarded and availability of data is also limited.

In order to have estimates of discard proportions of commercial or estimates of the proportion of the non-target fish killed by fishing we used an approach based on estimates of swept areas from fishing fleets (Pope *et al*., 2000) that does not require sampling of commercial by-catches and the required data on the distribution and abundance of non-target species are often available from surveys (Kunitzer *et al*., 1992; Knijn *et al*., 1993).

For a region like that covered by the NSRAC, the direct effects of fishing expressed as the mortality/removal of a specific ecosystem component in a spatially disaggregated system is determined by:

1. The abundance of that component in each spatial unit. For this we applied a slightly modified version of the method developed by (Sparholt 1990). We followed (Sparholt 1990) in that we estimated the abundance of non-target species by combining MSVPA-based abundance estimates of target species with survey catches that include both target and non-target species but improved the method in that we retained a size component and the spatial structure inherent to the survey data.
2. The frequency with which each spatial unit is fished with a specific type of gear. This was estimated using available quantitative data or informed estimates of relevant fishing parameters (e.g. fishing speed) and gear characteristics (e.g. width of the gear).
3. The impact of the single passing of the gear expressed as the proportion of that component removed. The direct effect of a fishery on a species is determined according to (Pope et al. 2000) but the model is further improved with regard to the assumption of a 100% catch efficiency. The interaction between fish and bottom trawls is a complex issue and determined by fish behaviour in relation to gear characteristics, making the catch efficiency of a gear hard to quantify (Wardle 1988; Dickson, 1993). Based on the available literature (Weinberg et al. 2002; Engås & Godø 1989) we estimated catch efficiency using information on (1) the positioning in the water column, (2) herding, (3) escape below footrope and (4) retention in the net. The latter is to a large extent determined by the mesh-size. For this we assumed the most commonly used mesh-size to apply for the whole fleet, i.e. 100 mm for otter trawl, and 80 mm for beam trawl.

For estimates of fishing impact at the level of the NSRAC this was integrated over all spatial units.

The direct effects of fishing on the North Sea fish community are estimated for the following components of the fish community:

- Small (10-25 cm) demersal non-target fish
- Large (> 25 cm) demersal non-target fish
- Elasmobranchs (> 10 cm)

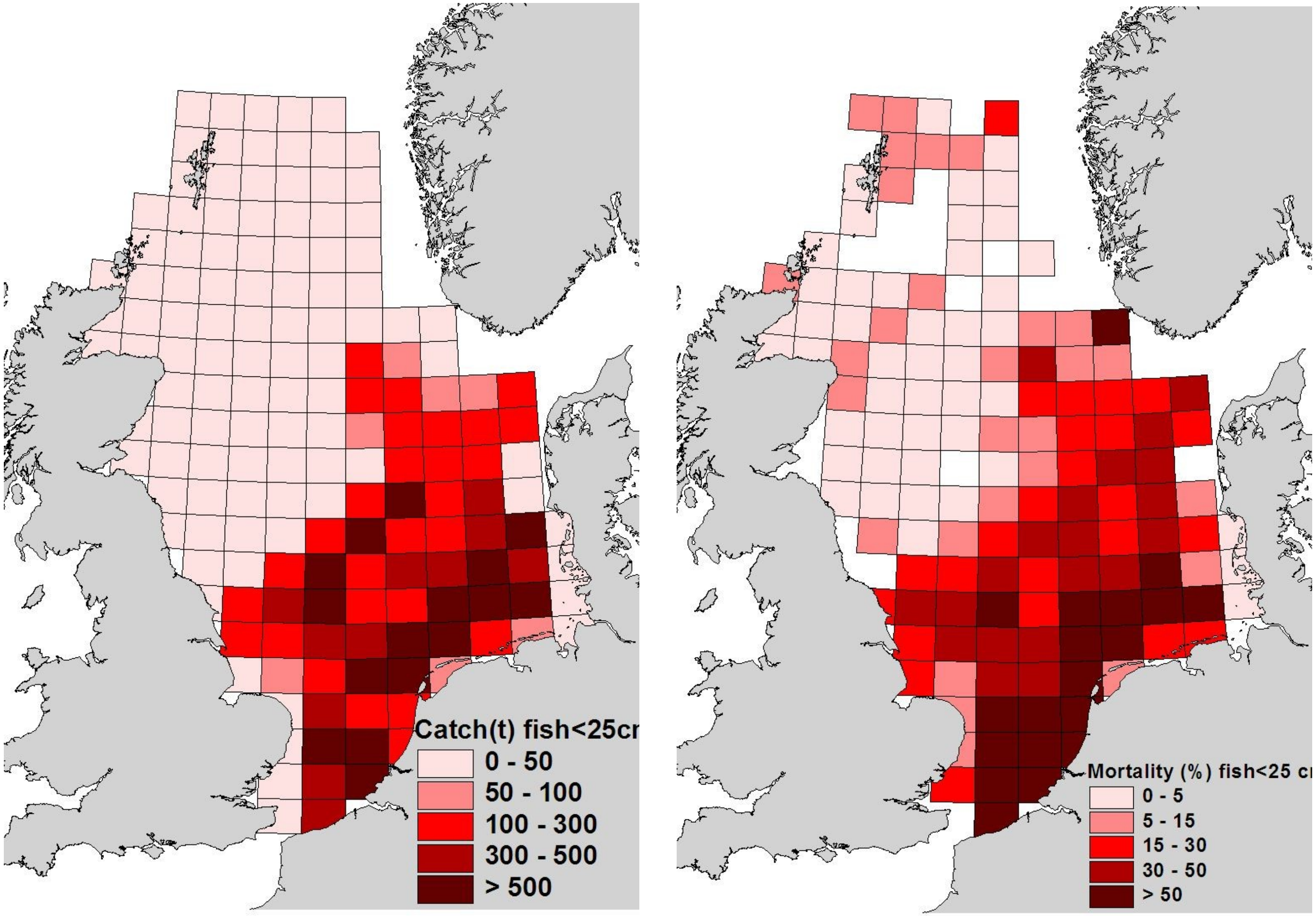


Figure 3.5.5.2. Fishing impact on non-target demersal fish smaller than 25 cm expressed as total catch in tonnes and the mortality as percentage of the total biomass present in that rectangle.

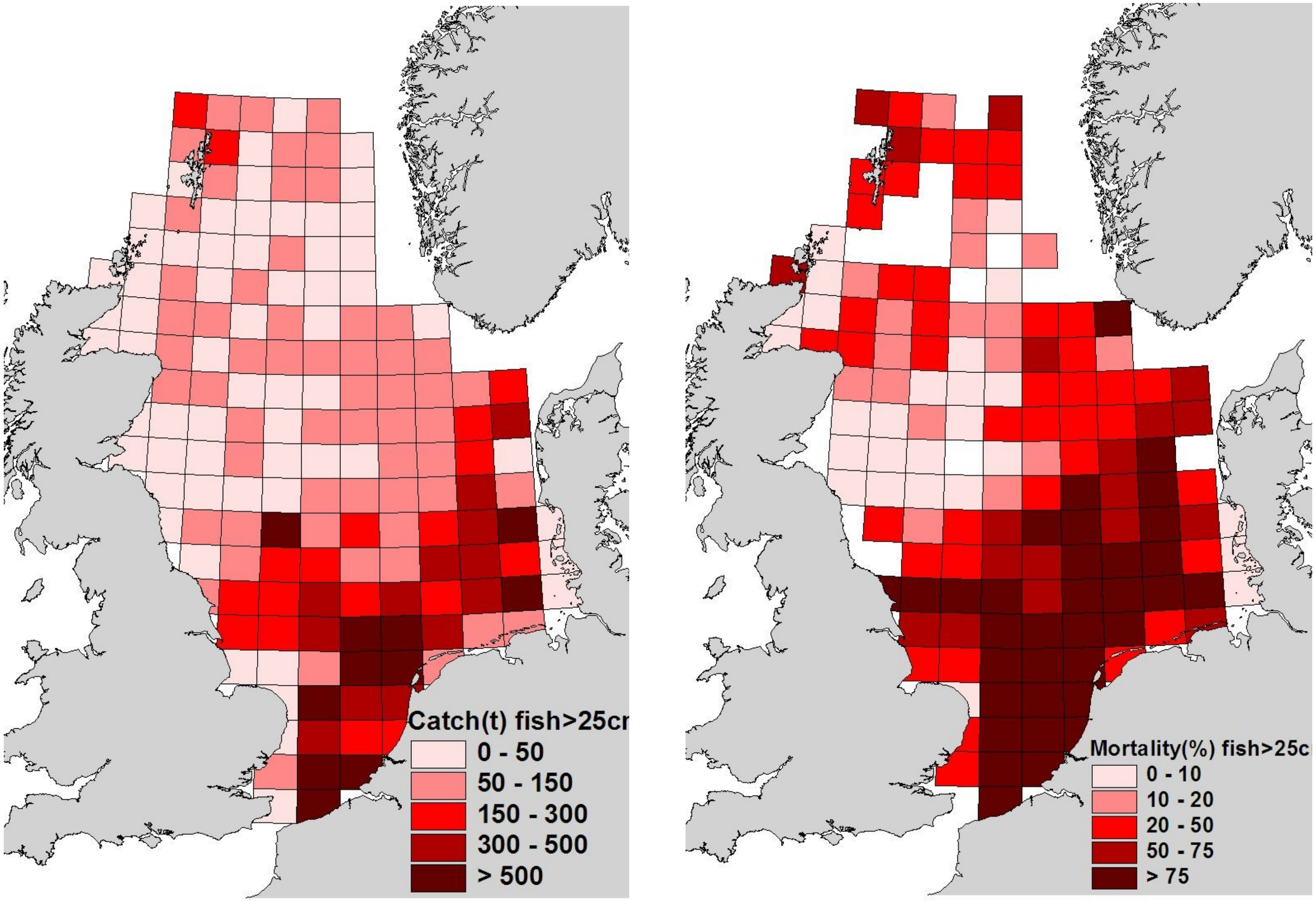


Figure 3.5.5.3. Fishing impact on non-target demersal fish larger than 25 cm expressed as total catch in tonnes and the mortality as percentage of the total biomass present in that rectangle.

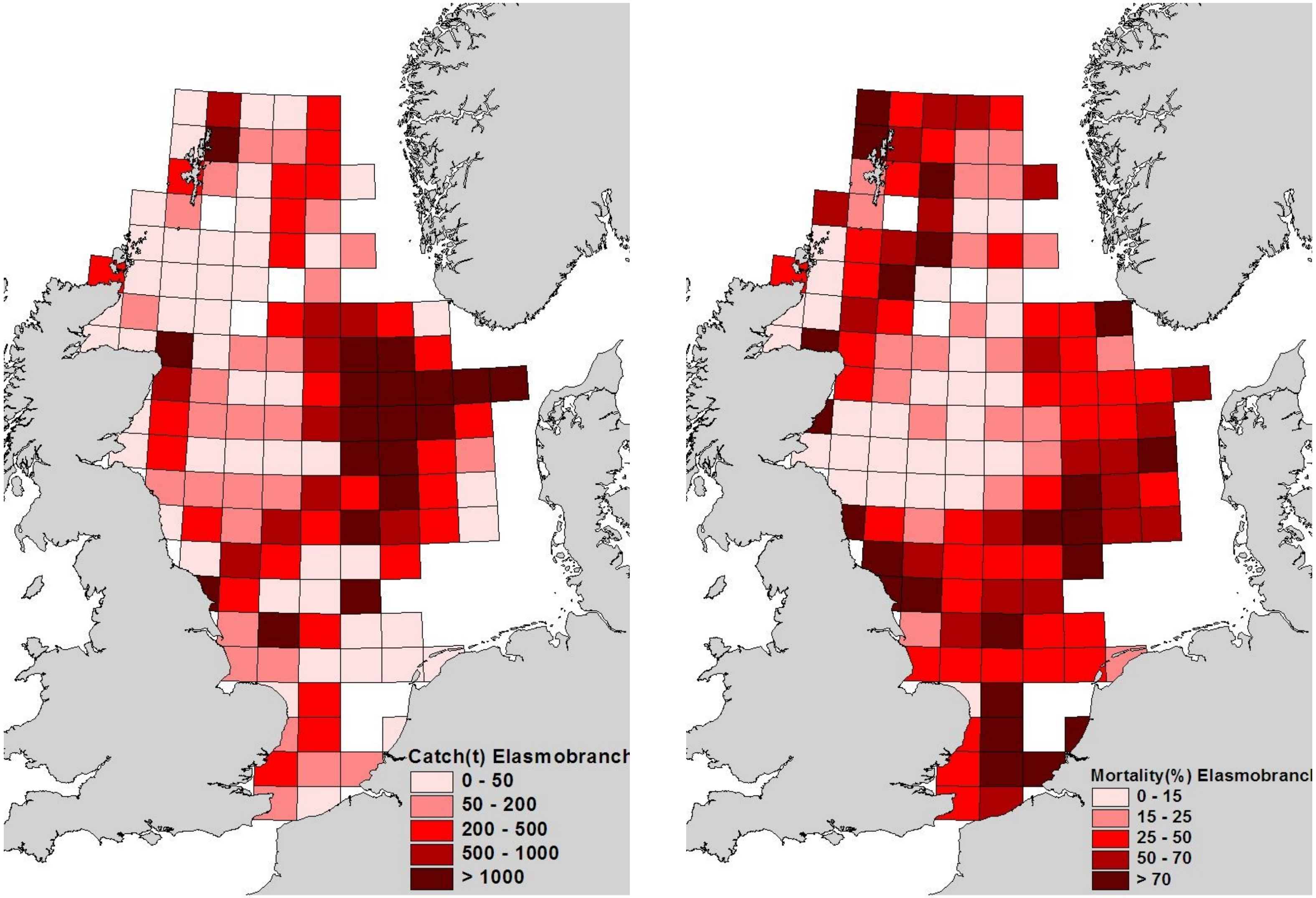


Figure 3.5.5.4. Fishing impact on elasmobranchs (sharks and rays) expressed as total catch in tonnes and the mortality as percentage of the total biomass present in that rectangle

For these components the fishing impact by two of the métiers (i.e. otter trawl and beam trawl) considered in this section is expressed as the biomass in tonnes that is caught and discarded as well as the proportion this makes up of the total biomass present (Figure 3.5.5.2-3.5.5.4). The rectangles for which information is missing are rectangles in which this was not caught by the survey.

	COD	HAD	PLA	SOL	WHI
Comparison datasets					
Landings MAFCONS (tonnes)	38064	36158	68890	17965	14505
Landings STECF (tonnes)	42536	37352	55268	18967	17302
Simulation model output					
Landings	19760	17645	51359	12121	3739
Discards	887	5718	17784	901	607
Proportion (%) of the catch discarded	18	23	42	24	12
Mortality (%) of the present biomass	38	7	51	47	5

Table 3.5.5.1. Actual and modelled information on fishing effects integrated for the NSRAC area (only ICES area IV).

This table shows that when aggregated over all rectangles and countries the total landings are of the same magnitude. No comprehensive comparison of the datasets was done and the observed differences may at least be partially caused by the countries included in each set. The MAFCONS dataset consists of Germany, UK, Netherlands and Norway. The STECF dataset consists of Belgium, Denmark, UK (Scotland and England), France, Germany, Ireland, Netherlands and Sweden.

With the current parameterisation of the simulation model the modelled landings are underestimated for all species, for the species caught by otter trawl more so than those caught by the beam trawl. This implies that the estimates of discards are probably also underestimated but this could not be validated. The relative measures of fishing impact (i.e. “Proportion (%) of the catch discarded” and “Mortality (%) of the present biomass”) are not necessarily affected by this.

Although with the current parameterisation of the model the estimates are clearly biased, this exercise does show how available information may be combined in models that simulate impacts of fishing on ecosystem components for which actual measurements are often not available.

The above results show that in the NSRAC area there are considerable spatial differences in the landings and the proportion discards of different commercial species which are partly related to differences in distribution of fishing métiers that differ in their impact on specific ecosystem components. This type of information allows management measures directed at specific fleets or areas in order to achieve reductions in bycatch of certain commercial species, the protection of vulnerable non-target species like the elasmobranchs or targets pertaining to the amount of discards available for consumption by other ecosystem components like seabirds (see section 3.6.2).

3.5.6 Marine mammals

Fishing affects marine mammals directly through bycatch. In some cases the scale of this bycatch is non-sustainable for some marine mammal populations.

In the North Sea, bycatch of harbour porpoises in bottom-set gillnets has been shown to be likely to having an effect at a population level. While bycatch does occur in other fisheries, current evidence is that it is not at the same scale as in gillnets. EU Fisheries management has addressed these issues through Regulation 812/2004 and there is insufficient information yet to know whether these management actions are effective. Fisheries can also catch seals. This bycatch has not been researched in the North Sea, but is not believed to be significantly affecting the populations in the North Sea. The group did not have sufficient information to comment on Norwegian bycatches or any relevant management actions.

3.5.7 Seabirds

Fishing affects seabirds directly through bycatch. Although there have been few dedicated studies in the North Sea, it is believed that bycatch is generally sustainable for seabird populations.

In the North Sea, bycatch of seabirds has been recorded in gillnets in the Kattegat, inshore fixed gear off Scotland and in long-lines in the far north of the North Sea. There have been no recent records from the Kattegat, and most fixed net fisheries off Scotland have closed (to conserve salmon stocks). A re-examination of Kattegat fisheries and a study of bycatch in the northern North Sea and adjacent areas (where fisheries affect the same population of seabirds) is needed in order to understand the scale of any current effects and therefore any possible management measures.

3.6 Indirect effects of fishing on the North Sea ecosystem components

3.6.1 Small mesh fisheries

There are two main indirect effects of the small mesh industrial fisheries, the direct removals (the catch and by-catch) and the ecological consequences of these for ecologically dependent species. The concern is that the scale of the small meshed fisheries may lead to competition between these organisms and the fishery for the same resources.

3.6.1.1 Direct removals (catch and by-catch)

The impact on the eco-system of the catches and by-catches in the North Sea small meshed fisheries was reviewed by WGECO at the meeting in 2003 (ICES, 2003a) using data up to and including 2001. Here we update this review with the latest available information about the catches in the North Sea small meshed fisheries.

The catches and species composition in small meshed fisheries in the North Sea is presented in Table 3.6.1.1.1 with data up to and including 2004. The table shows a sharp decrease in catches of the target species in the North Sea small meshed fisheries from 2002 to 2003. The catches in 2004 were on a similar low level as in 2003. This decrease is due to a large decrease in effort and landings in the sandeel fishery caused by a large decrease in the size of the sandeel stock (see ICES 2006a). This low level of both stock and landings was also seen in 2005, and the sandeel fishery was closed at the end of the 2005 fishing season to protect the stock from overfishing. Both the size of the Norway pout stock and the landings in the Norway pout fishery has been decreasing since 2000, and in 2005 the fishery was closed to protect the stock from overfishing. In contrary to the decrease seen for sandeels and Norway pout there has been no decrease in the landings in the blue whiting and sprat fisheries in the latest years (ICES 2005b, c and 2006b).

The decrease in total landings has meant that there has been no increase in total by-catch in the small meshed fisheries since 1997. However, the decreasing trend, seen in the by-catch of the gadoids in recent years, has continued in the latest years, due to the decrease in effort in the sandeel and Norway pout fisheries and major decline in gadoid stocks.

In the review made by the WG in 2003 the effect of by-catch of haddock and whiting in the small meshed fisheries was assessed, and found to be minor and smaller than the impact of human consumption fisheries and natural mortalities. This year the WG tried to assess the effect of the by-catch of the other by-catch species in the small meshed fisheries in the North Sea. To make this assessment, information about the age, length and weight composition of the by-catch species is needed. In the report of the Working Group on the Assessment of the Demersal Stocks in the North Sea and Skagerrak there has been some reporting of such information for certain countries and years (see e.g. ICES 2006a). This year detailed information about the age, length and weight distribution from the Danish small meshed fishery was available to WGECO. The number of fish, sampled from the Danish small meshed fisheries for species composition by fishery inspectors, that have been length measured and aged are summarised in Tables 3.6.1.1.2 and 3.6.1.1.3. These data are important as these represent far the largest proportion of the small meshed fisheries in the North Sea. Except for haddock and whiting, for which information about industrial by-catch is included in the assessment, information about age and length distributions of the by-catch species is sparse. The majority of the "other species" group is comprised of the greater sandeel *Hyperoplus lanceolatus*.

Figure 3.6.1.1.1 and 3.6.1.1.2 shows the length and age distributions of by-catch species in the Danish small meshed fisheries, as an average for the period 1996 to 2005. The graphs show that the commercial by-catch species are generally below 20cm whereas non-commercial by-catch species are larger. Further, the largest fraction of the commercial by-catch species is age-0 with a small percentage of 1-group fish.

This year the WG decided to consider the effect of the by-catch of cod in the small meshed fisheries, as there has been a concern about this effect, and because some information about age distribution of cod in the by-catch was available. Due to the sparse information about the age, length and mean weight of cod in the small meshed fisheries, an average mean weight at age and age distribution was estimated for the time period 1996 to 2005 (Table 3.6.1.1.4). These estimates were used to break down total catches by year to catches by year and age. The mean weight at age, presented in Table 3.6.1.1.4, is smaller than those used by WGNSSK for the assessment of the North Sea cod (see ICES, 2006a). However, the small estimates presented in Table 3.6.1.1.4 reflect the poor condition of the fish landed for reduction and are considered more appropriate to use, when estimating the number of fish landed for reduction, than the estimates used by WGNSSK for the cod assessment.

Using the estimates of mean weight at age and age distribution in Table 3.6.1.1.4 the number of cod by year and age retained in the small meshed fisheries in the North Sea was estimated (Table 3.6.1.1.5). These estimates do not include larger fish counted against the human consumption quotas. The table shows that the number of 0-group cod retained as by-catch in the small meshed fisheries is at about the same level as the numbers discarded in the human consumption fisheries. Further, the number of age-1 cod and older retained as by-catch in the small meshed fisheries are small compared to the numbers retained and discarded in the human consumption fisheries. Taken into account the relatively large natural mortalities that have been estimated for 0-group cod (estimated to 4 and 3 for 2002 and 2003 respectively, see ICES 2005d) the overall conclusion is, that the small meshed fisheries in the North Sea has a small effect on the North Sea cod population. This conclusion should however be taken with some reservations. Firstly the estimates of mean weight at age and proportion of cod by age, used in the calculations, was estimated as an average over a relatively long time period. Both estimates are assumed to show a large variation over both time and space. Further, differences between fisheries should also ideally have been taken into account. The estimates of number of cod retained as by-catch in the North Sea small meshed fisheries were thus primarily produces to give the first impression about the effect of the small meshed fisheries on the cod population.

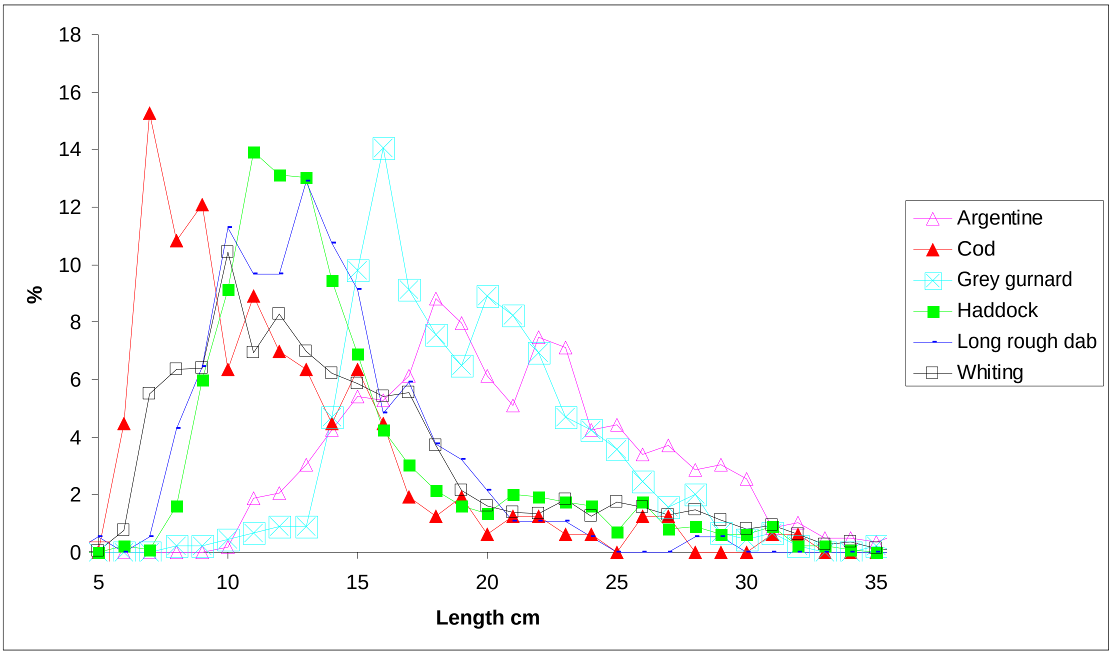


Figure 3.6.1.1.1. Length distributions of by-catch species in the Danish small meshed fishery in the North Sea. Average over the years 1996 to 2005.

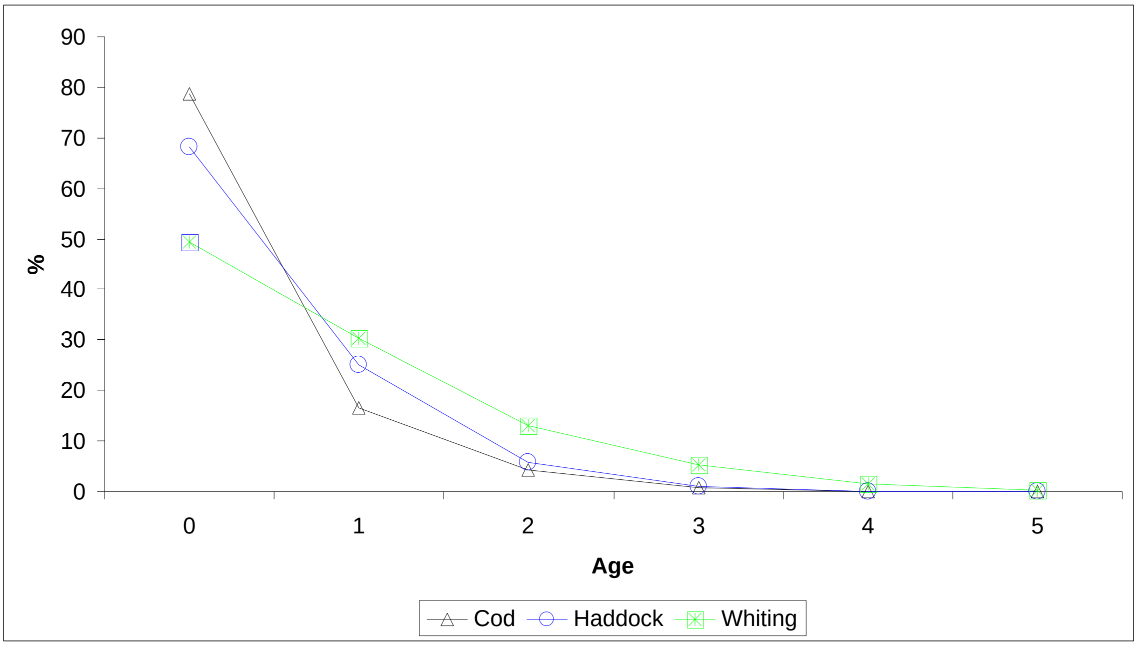


Figure 3.6.1.1.2. Age distributions of cod, haddock and whiting in the Danish small meshed fishery in the North Sea. Average over the years 1996 to 2005.

Table 3.6.1.1.1. Total catch (000 t) of main target species (sandeel, sprat, blue whiting and Norway pout) and by-catch species in the Danish and Norwegian small meshed fisheries of the species landed for reduction. From ICES (2006b).

	Target species	By-catch species																		
Year		Herring	Haddock	Whiting	Saithe	Cod	Macrerel	Horse macrerel	Trigla sp.	Dab	Smelt sp.	Long rough dab	Place	Hake	Poor cod	Ling	Witch	Silvery pout	Others	Total by-catch
1985	942,000	63,000	6,000	15,000	8,000	544	4	22,789	0	187	8,714	59	34	349	0	51	236	1,210	31,715	157,892
1986	1,075,000	40,000	3,000	18,000	1,000	710	534	16,658	888	3,209	5,210	718	119	165	68	1	132	729	3,853	94,994
1987	1,035,000	47,000	4,000	16,000	4,000	1,092	2,663	7,391	45,342	4,632	3,033	1,173	109	261	0	40	341	3,043	3,604	143,724
1988	1,110,000	179,000	4,000	49,000	1,000	1,404	6,414	18,104	5,394	3,781	1,918	946	372	242	5	39	44	2,494	3,670	277,827
1989	1,292,000	146,000	2,000	36,000	1,000	2,988	8,013	22,723	9,391	7,743	778	2,160	582	290	48	37	255	741	3,528	244,277
1990	824,000	115,000	3,000	50,000	8,000	2,948	5,212	14,918	2,598	4,706	2,801	1,673	566	429	121	13	251	476	3,154	215,866
1991	1,136,000	131,000	5,000	38,000	1,000	570	7,466	5,704	5,622	5,578	3,434	1,024	1,305	28	79	65	1,439	801	4,444	212,559
1992	1,365,000	128,000	11,000	27,000	0	1,044	4,631	6,651	4,209	3,986	2,024	1,694	218	359	111	10	195	0	4,553	195,685
1993	922,000	102,000	11,000	20,000	1,000	1,052	4,386	6,169	1,593	4,871	2,874	1,428	128	109	36	28	246	0	4,106	161,026
1994	1,233,000	40,000	5,000	10,000	0	876	3,576	4,886	1,139	528	2,209	529	143	10	0	0	40	0	5,141	74,077
1995	1,434,000	66,000	8,000	27,000	1,000	955	2,331	2,746	2,091	1,028	292	617	33	0	9	0	0	0	5,158	117,260
1996	1,057,000	39,000	5,000	5,000	0	366	2,019	2,369	897	1,065	3,101	339	90	3,625	30	0	97	7	50	63,055
1997	1,362,000	15,000	7,000	7,000	3,000	1,688	3,153	3,332	2,618	2,662	2,604	1,411	73	2,364	181	31	394	248	749	53,508
1998	1,231,000	16,000	5,000	3,000	3,000	1,281	1,934	2,576	1,015	6,620	5,205	2,229	91	33	261	31	860	248	5,405	54,789
1999	1,030,000	23,000	4,000	5,000	2,000	532	2,728	5,116	2,566	4,317	3,580	1,272	88	211	922	125	437	387	17,931	74,212
2000	1,120,000	24,000	8,000	8,000	6,000	383	2,443	5,312	1,343	441	333	493	64	231	518	19	154	532	8,927	67,193
2001	1,101,000	21,000	6,000	7,000	3,000	192	1,749	1,159	2,293	1,441	397	431	56	167	0	49	246	942	301	46,423
2002	1,126,000	26,000	4,000	8,000	8,000	29	1,260	2,338	1,071	321		112	51	6	196	0	58	459	2,226	54,127
2003	635,000	16,000	1,000	3,000	8,000	49	2,549	5,791	847	596	1,376	208	28	301	5	42	437	993	4,888	46,110
2004	636,000	19,000	1,000	2,000	7,000	44	6,515	10,272	1,101	386	786	174	1	423	91	169	286	1,550	6,953	57,751
Average	1,083,300	62,800	5,150	17,700	3,300	937	3,479	8,350	4,601	2,905	2,667	935	208	480	134	38	307	743	6,018	120,618

Table 3.6.1.1.2. Number of fish, that have been sampled from the Danish small meshed fishery in the North Sea and length measured, by year and species. Industrial species are here sandeel, sprat, Norway pout, blue whiting and herring.

Year	Industrial species	Argentine	Cod	Grey gurnard	Haddock	Horse mackerel	Lemon sole	Long rough dab	Plaice	Saithe	Whiting	Other species	Witch	Total
1996	27939		62	16	264	162		22	3		98	1493		30059
1997	17590		15	12	96	838		23			88	1133	2	19797
1998	23595	176	37	26	47	416	1	76	1		307	2379	16	27077
1999	32084	193	1	150	356	125		13			413	2037	2	35374
2000	27891	55	2	22	249	168		12			619	1608	5	30631
2001	12944	81	8	49	37	18		20			171	10355	8	23691
2002	25035	53	16	43	17	54		15	2	1	130	6686		32052
2003	18210	49	4	52	214	215		14	1		165	5121		24045
2004	15505	30		34	2	755		1		1	47	4235		20610
2005	16278	28	12	48		288		3	1		94	2097		18849
Total	217691	665	157	452	1282	3039	1	199	8	2	2132	37263	33	262924

Table 3.6.1.1.3. Number of fish, that have been sampled from the Danish small meshed fishery in the North Sea and aged, by year and species. Industrial species are here sandeel, sprat, Norway pout, blue whiting and herring.

Year	Industrial species	Argentine	Cod	Grey gurnard	Haddock	Horse mackerel	Lemon sole	Long rough dab	Plaice	Saithe	Whiting	Witch	Other species	Total
1996	7579		60	0	257	0		0	0		96			7992
1997	4433		15	1	89	0		0			72	0	0	4610
1998	6174	0	36	0	35	0	0	0	0		240	0	0	6485
1999	8543	1	1	0	276	0		0			235	0	0	9056
2000	6558	0	2	0	128	0		0			345	0	0	7033
2001	3179	0	8	0	37	0		0			59	0	1	3284
2002	2152	0	10	0	10	0		0	0	1	99		0	2272
2003	7063	0	2	0	4	40		0	1		124		0	7234
2004	6819	0		0	2	122		0		0	37		0	6980
2005	5463	0	11	0		75		0	0		91		0	5640
2006	153												0	153
Total	58116	1	145	1	838	237	0	0	1	1	1398	0	1	60739

Table 3.6.1.1.4. Estimates of mean weight at age (g) and % of fish by age for cod caught as by-catch in the Danish small meshed fisheries, calculated as an average over the time period 1996 to 2005.

Age	0	1	2	3
Mean weight (g) at age	13.0	59.9	167.2	155.0
% in numbers by age	78.62	16.55	4.14	0.69

Table 3.6.1.1.5. Estimates of number of cod caught as by-catch in the North Sea small meshed fisheries, together with estimates of number of cod retained and discarded in the human consumption fisheries (the last two estimates are from ICES 2006b).

Year	Age	Ind. b.	Discard	H.comsumption	Total
2002	0	738			738
	1	155		6139	6294
	2	39		6170	6209
	3	6		10810	10816
	4			1849	1849
	5			213	213
	6			273	273
	7			43	43
	8			29	29
	9			12	12
	10			5	5
	11				0
2003	0	1246	777		2023
	1	262	7328	364	7954
	2	66	8454	7871	16391
	3	11	1065	2923	3999
	4		70	2620	2690
	5		14	442	456
	6		2	50	52
	7		1	49	50
	8		0	13	13
	9			7	7
	10			3	3
	11		0	1	1
2004	0	1119	989		2108
	1	236	8907	1496	10639
	2	59	4086	3602	7747
	3	10	911	4274	5195
	4			1279	1279
	5			856	856
	6			121	121
	7			31	31
	8			19	19
	9			7	7
	10			2	2
	11			0	

3.6.1.2 Impacts resulting from loss of forage fish

3.6.1.2.1 Sandeels

Many seabirds species are highly dependent on sandeels for prey, and for many of these, and for the species that are not sandeel specialists, other small fish species targeted by the small meshed fisheries, such as sprats and Norway pout, constitute some of the main alternative prey (Furness 2002; Wanless et al. 1998). The preference for sandeel and sprat prey is explained by the higher calorific value of these two species (Hislop et al. 1991). Although the precise mechanism remains unclear, seabird breeding success has repeatedly been linked to the abundance of sandeels (Monaghan, 1992; Hamer et al. 1993; Rindorf et al. 1978). Even where environmental conditions have been shown to strongly influence breeding success, additional detrimental fishing effects have been demonstrated (eg. Frederiksen et al. 2004; Scott et al. in press). Since sandeels appear to be particularly important to seabirds during the breeding season, it is important to consider the spatial and temporal overlap of seabird and fishery distributions at this time. The fishery generally takes place in quarters 2 and 3, coinciding with the seabird breeding season, when energetic demands are particularly high. Breeding seabirds tend to feed close to their colonies (Furness & Tasker 1997; Daunt et al submitted) and in some areas fisheries overlap spatially with these foraging areas (Wright & Begg 1997; ICES 2003a). If local prey resources are depleted by fisheries, forcing seabirds to forage over longer distances, this will result in increased foraging energetic costs (Krebs & Davies 1993). The main sandeel fisheries in the North Sea occur relatively far offshore in the central and southern North Sea. However, on occasion some relatively large-scale fisheries have taken place relatively not far off the northern and eastern coasts of Scotland, close to some of the largest seabird colonies in the North Sea region. Thus, the potential for competition between seabirds and small meshed fisheries certainly exists at some localities.

It is difficult to determine the effect of the sandeel fishery and sandeel biology on seabird population dynamics. Seabirds can be considered indicator species, their condition relating to the status of the health and productivity of coastal and marine systems (Furness and Camphuysen, 1997) but their condition is not a direct proxy of the abundance of fish stocks. Since 1997, two EC Studies (ELIFONTS {Effects of Large-Scale Industrial Fisheries on Non-Target Species} and IMPRESS {Interactions between the Marine environment, PREdators and prey: implications for Sustainable Sandeel fisheries}), along with an intervening Scottish FRS (Fisheries Research Services) research project, have investigated the diets and breeding success of common guillemots (*Uria aalge*), European shags (*Phalacrocorax aristotelis*), and black-legged kittiwakes (*Rissa tridactyla*) at an important seabird colony, the Isle of May, in the Firth of Forth. Data were also collected on local hydrograph conditions and the abundance, distribution, behaviour and size/age composition of the local sandeel population. These studies showed that relatively small changes in the timing of peak sandeel availability in June were a major determinant of seabird breeding success. The kittiwakes were especially vulnerable to changes in sandeel availability as they did not switch to prey on other species (e.g. Rindorf *et al.*, 2000). The timing of two events in the sandeel lifecycle appears to be critical for the success of bird populations. These are: 1) the onset of burrowing behaviour and 2) the arrival of 0-group fish on the seabirds' feeding ground, both of which are primarily driven by environmental factors. However, despite the clear role of the environment in influencing feeding opportunities for kittiwakes, fisheries operating on local sandeel grounds also have a significant additional negative effect (Frederiksen et al. 2004; Scott et al. in press).

Fishing down local 1+ sandeel populations might be considered to pose a threat to local spawning stocks, and so potentially present a risk to future recruitment. However, the evidence available suggests the opposite. 0 group abundance appears to correlate negatively with the abundance of 1+ sandeels, implying some form of density dependent interaction (Arnott & Ruxton 2002; Furness 2002). The ELIFONTS/IMPRESS projects studying sandeel population dynamics on the Wee Bankie sandeel grounds off the Firth of Forth, SE Scotland, note that 0 group recruitment was high in the first year of the closure off the sandeel fishery off the east Scottish coast, despite low biomass levels of 1+ sandeels in the preceding year (Greenstreet et al., submitted).

Industrial feed fish species are present in the diet of harbour porpoise *Phocoena phocoena*, bottlenose dolphin *Tursiops truncatus*, white-beaked dolphin *Lagenorhynchus albirostris* and minke whale *Balaenoptera acutorostrata* in the North Sea (Borjesson *et al*., 2003). The proportion of these fish reported in the diet varies by season and by geographic location. In Scottish waters, sandeels may constitute up to 58% by weight of the stomach content in harbour porpoises, other feed fishes, sprat and Norway pout, can be less than 1% by weight. In Kattegat and Skagerrak, other feed fish (mainly sprat and herring) constitute 13% by weight of the stomach content in juveniles and 10% by weight in adult harbour porpoise stomachs (Borjesson *et al*., 2003). Sandeels can form more than 80% to the diet by weight of minke whale in the North Sea, but further north (and possibly when herring are more abundant), the diet of minke whales can be dominated by herring (Olsen and Holst, 2001).

As with other ‘opportunistic’ predators, differences in the diet composition reflect the local occurrence of potential prey. Unlike seabirds which predominantly target the smaller (0-, 1-group) sandeels, marine mammals can take the older and larger fish. Sandeel fisheries may therefore impact marine mammal populations by altering their food supply in certain areas. It is therefore important to consider the local availability of sandeels to cetaceans, and their ability to switch to other prey if the stocks are depressed, when assessing the effects of sandeel fisheries on marine mammals. A direct link of fishing for sandeels to cetaceans, however, has yet to be demonstrated in any population.

Sandeels (mainly *Ammodytes marinus*) can form an important part of the diets of both grey *Halichoerus grypus* and harbour *Phoca vitulina* seals, particularly in the summer months (Prime & Hammond 1987; Thompson *et al* 1996; Tollit *et al* 1997). Both seal species have a wide range of foods and there is little to suggest that either of these species is particularly dependent on the fish targeted by the small mesh fisheries (Hall et al 1998; Hammond et al 1994; Pierce et al 1991; Prime & Hammond 1990; Tollit & Thompson 1996). Recent population growth trajectories do not suggest that they are ‘food limited’ in any major way (Harwood, 1999).

3.6.1.2.2 Sprat

There is some evidence for competition between the sprat fishery and wintering Common guillemots (*Uria alga*) leading to late winter mortality (Jensen *et al*. 1994; Blake, 1984). The sprat effort map (figure 3.4.4.3.1) indicates that the highest landings are taken adjacent and within the Wadden Sea SPA. Competition between small meshed fisheries may occur and local seabird breeding populations may occur. Guillemots at the Isle of May colony in the Firth of Forth, SE Scotland, consume a high proportion of sprats in their diet, despite their relative scarcity in the region, and despite a considerable increase in the abundance of sandeels at the nearby Wee Bankie sandbanks (Daunt et al submitted; Greenstreet et al submitted).

3.6.1.2.3 Norway pout

The fishery for Norway pout is unlikely to affect seabirds by affecting their food supply. The ecosystem interactions of this fishery on marine mammals has not been evaluated.

3.6.1.2.4 Blue whiting

Blue whiting are rare in seabird diets in EU waters. It seems unlikely that there would be a great indirect effect on seabirds from the fishery. The ecosystem interactions of this fishery on marine mammals has not been evaluated.

3.6.1.2.5 Impacts of changes in forage fish for fish consumers

Although previous analysis of the food web energy flow through to fish in the North Sea has suggested that demersal piscivores may be the most food limited group (Greenstreet et al 1997), the role of food shortage arising as a consequence removals by the small mesh fisheries has not received much attention at the North Sea scale. Where data are available at the local scale no "competitive interaction" between the fisheries and piscivorous predators has been apparent. Despite closure of the sandeel fishery off the east coast of Scotland, and the resultant immediate increase in the local biomass of sandeels (Greenstreet et al submitted), gadoid populations in the area have continued to decline and the proportion and consumption rates of sandeel in their diets have remained unchanged (Greenstreet 2006). Recent increases in the abundance of pelagic piscivorous fish also suggest that they have in no way been limited by fisheries induced reductions in the abundance of their prey (Heath 2005).

3.6.2 Ecological consequences of discarding

Most fishing operations catch and kill organisms beyond those that are commercially valuable. This portion of the bycatch is usually discarded back into the sea. The fate of this discarded material varies depending on its composition, amount and location.

Composition will vary with type of fishery – a smaller meshed net, for e.g. brown shrimp, will catch proportionately more small, unwanted fish than a large mesh net, if used in the same area. The morphology of discarded fish will vary – with more flatfish being discarded in the southern North Sea than the northern North Sea; in some cases (rare in the North Sea), discards are minced before discharge. Amount varies greatly by fishery, with little or none from industrial small-mesh fisheries, occasional large quantities in one place caused by slipping of an entire catch in the pelagic fishery, moderate amounts of undersized fish and offal from demersal trawl fisheries (the amount of undersized fish discarded depending on the size profile of individual stocks of fish in any year) and relatively large quantities of undersized fish and other biota from shrimp fisheries.

These factors also effects the consumers of the fishery waste. Camphuysen *et al*. (1995) used research vessels to study seasonal patterns in the spatial distribution of scavenging seabirds in the North Sea and to study the attraction of fishing vessels for these birds. The selection and consumption of discards by seabirds was quantified during sessions of experimental discarding. The spatial distribution of fishing vessels in the North Sea and discard practices were investigated and the results were used to analyse the attractiveness of different fisheries for seabirds.

Eight species of seabird utilised fishery waste on a large scale, at least during part of the year. Consumption rates by seabirds, which were higher in winter than in summer, ranged from 95% for offal, to 80% for roundfish, 20% for flatfish and 6% for benthic invertebrates. All length classes of discards which occur normally in commercial fisheries can be consumed by seabirds. The median length of experimentally discarded roundfish consumed by seabirds ranged from 15cm in a small species such as the black-legged kittiwake to 25cm in northern gannets and great skuas. Northern fulmars and black-legged kittiwakes were specialised feeders on offal. Discards size selection by different species of seabirds overlapped, leading to inter-specific competition. Many discards were stolen from smaller birds by larger species.

The fate of discards not consumed by seabirds has not been directly investigated, though the results of investigations of the stomach contents of fish revealed that discarded fish and offal are consumed by dabs and probably other fish. It is assumed that most discards reaching the seabed are consumed by scavenging fish and invertebrates.

Discarded waste from fishing vessels thus affects the ecosystem by subsidising the diets of scavenging species. The nature of discarding also means that this subsidy will move higher in the water column, or out of it, compared to the location of the live organisms before discarding. The distortion caused by the increase in the numbers of scavengers may have secondary effects when the amount of food discarded decreases (e.g. due to smaller fish stocks or reduced fishing effort). In Shetland, the reduction in discarded fish (and in the local sandeel stock) has meant that great skuas have switched diets to direct feeding on other seabirds, with consequential decreases in these avian prey populations.

3.7 Community level metrics of the effects of fishing on ecosystem properties

In past meetings WGECO has discussed and evaluated many possible indicators of the effects of fishing on marine communities and aggregate or emergent ecosystem properties. Much of this past work is summarized in Rice (2005). In those evaluations WGECO identified three classes of community properties that might be of interest to those wishing information on status of trends of marine communities, and those responsible for management of human activities affecting those communities. These classes were biodiversity, functional relationships among ecosystem components, and biogenic and abiotic structural features of the marine habitat. Each of these classes, in turn, has a number of subcomponents that have been discussed in the scientific literature, and each of these has many possible indicators.

WGECO's reviews and evaluations have concluded that all the available indicators in some of these classes lack key qualities that are desirable in useful indicators of status and trends in community properties, and even more lack properties that are desirable in indicators to be used in guiding management. In this exercise only indicators judged to be potentially useful in our past reviews were considered for assessment of the overall effects of fishing on the North Sea.

For this exercise all reported indicators refer only to the fish community or their habitats. Moreover, indicators are usually calculated only for the portion of the fish community usually sampled by research survey gears and sometimes only as reflected by analyses of data sets from commercially exploited species. This is not done because WGECO considers that the fish community has any special significance relative to other major components of the marine ecosystem such as zooplankton or benthos. Rather this is done because of severe data limitations to assess the trends over the past decades in most other ecosystem components, and because it is the fish community that is likely to be impacted most directly by fishing. A number of scientific studies have found trends in other ecosystem components such as pelagic zooplankton (Beaugrand et al. 2002, Edwards and Richardson 2004, Richardson and Schoeman 2004). However, generally these studies report either little ability to partition with confidence the indirect effects of fishing from those of many other top-down and bottom-up processes, or else highlight some major environmental event as an important contributor to the observed trends.

Below we extract major findings from recent publications that report status or trends for the North Sea in several indicators that have been identified as having potential to reflect ecosystem effects of fishing. However, in only a very few cases have researchers attempted to link patterns in the indicators to fisheries as the specific cause of any observed trends. In the other cases, if the reported patterns are interpreted in the first instance as due completely to fishing, then we have a worst-case scenario, and the true effects of fishing may actually be less.

3.7.1 Biological Diversity

3.7.1.1 Abundance

WGECO did not recommend any particular indicator of the total abundance of the fish community, but noted short-comings in all candidates. Aggregate landings is a possible candidate indicator of total abundance, but fishing mortality on many target species has varied as much as two-fold or even more over the past 25 years, and discarding practices have been highly variable over time in some fisheries (ICES 2005a). Hence patterns from landings may be misleading if species contributing a substantial portion of the total biomass or abundance showed marked changes in exploitation or discarding. Trends in aggregate survey catch numbers or biomass across all species give another indicator of trend, but such indicators are affected by differential catchability of species in the survey. Trends in total numbers or biomass across all predator and prey species as estimated by Multispecies VPA gives another indicator of trends in total abundance over time. This indicator is not affected by differential catchability in fishing gears, but does not represent all species in the community. Rather, a box called “other prey” which includes all fish not represented explicitly in MSVPA as well as all other non-fish sources of food is allowed to vary as needed to ensure that all predators receive their feeding ration. Still, by taking direct account of predation losses in the system, trends in total abundance and biomass of the MSVPA estimates are more informative of trends in total fish community attributes than the sum of biomasses across all single-species SPAs.

All three types of indicators considered below are imperfect, but are considered most likely to reflect variation in the part of the community impacted most directly by fishing, and hence each can be considered to provide some information on fishing effects. Unless some species not represented in MSVPA nor sampled well in the survey gears, are more strongly affected by fishing than the target species in fisheries (which may be the case for some elasmobranchs and species with similar life histories) *and* have key roles in ecosystem function, these metrics here will be worst-case scenarios for fishing effects on total abundance or biomass of the fish community.

Heath (2005; fig 5) presented trends in annual landings in 10^6 tons from 1973 to 2000. These trends are disaggregated by trophic guilds. Disaggregated they show a fairly steady decline demersal piscivores since the mid 1970s (and especially in the late 1980s) and a greater than 7-fold drop in pelagic piscivores between 1973 and 1980, followed by a more than 8-fold increase to 1992, and subsequent irregular decline by about 40%. Benthivores show a steady decline through the 1970s to about 60% o f the 1973 value, an increase to more than 125% of the 1973 value through the 1980s, and a 50% decline through the 1990s. However, the *total* biomass in the system, as estimated by landings, has been dominated by planktivores which have varied without trend over the whole period. Aggregated across the trophic guilds in Fig 5 of Heath, there was an initial decline in total landings through the 1970s, when all groups except benthivores were at their highest observed landings, but thereafter the losses in demersal biomass were approximately compensated for by increases in the other groups. Heath concludes that only the decrease in the demersal piscivores can be attributed to fishing. Otherwise, although fishing affected individual species at various times, other populations responded in ways that meant the overall community biomass did not experience any major directional change due to fishing.

A paper currently in journal review reports a downward trend in biomass since the early 1970s based on North Sea survey data, but these results are hot available yet. However calculating aggregate numbers is a preliminary step in calculating survey-based size spectra of a system, and there are many such spectra published for the North Sea (see Section 3.7.2.1). Daan et al. (2005) provide a method for estimating the “height” of a size spectrum separately from its slope, and the height is a surrogate for the total number of fish underneath the slope. Two of the three surveys examined show a statistically significant decline in height of the North Sea size spectrum between 1977 and 2000, and the third also shows a decline, although it is not significant. These changes in "height" of the spectrum cannot readily be converted into total changes in numbers of fish in the community because the slopes of the spectra do change as well. Moreover, the two survey trends that change significantly are reduced by over a third in one case and less than a sixth in the other. However, in both cases the changes, corresponding to a reduction in total numbers of fish in the North Sea system, were linked directly to fishing.

The pattern of aggregate biomass and abundance of fish represented in MSVPA since 1963 (figs 3.7.1 and 3.7.2 taken from MSVPA ICES 2005d) indicate that both indicators reached a nadir in the mid 1980s, and have increased somewhat since. The minimum was proportionately lower in numbers (>50% less that the values in the early 60s) than in biomass (<40% decrease over the same period), but the subsequent increase has also been greater as well (nearly 40% and possibly still rising for numbers; no more than 20% increase for biomass and varying without trend since ~1987), and possibly even declining again in most recent years.

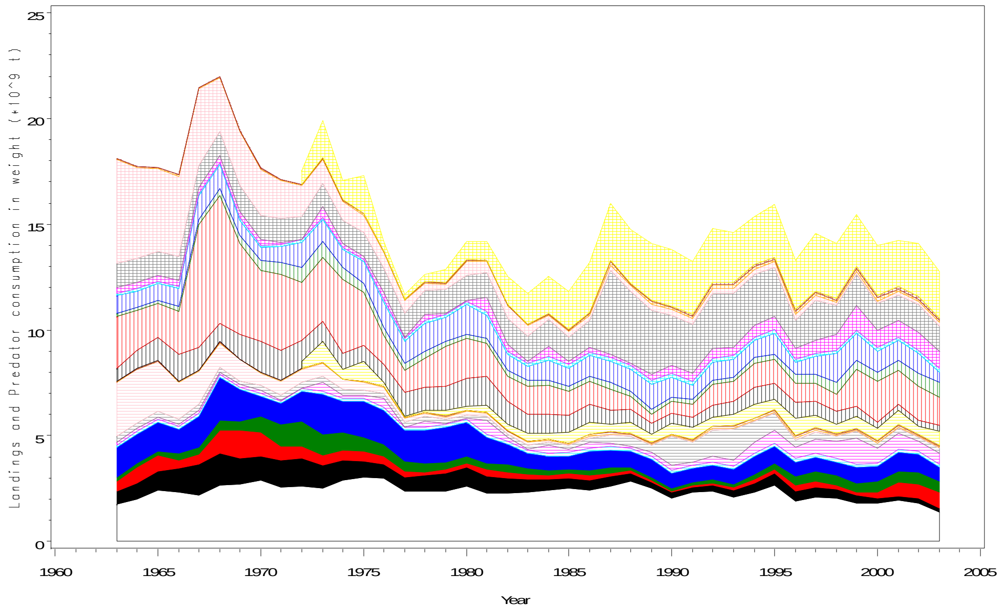


Figure 3.7.1 : Time trends in the biomass of fish landed and consumed by predators; results from the key-run (from ICES 2005d – SG Multispecies assessment for the North Sea).
Colours indicate the predator species; hatching the status of predator and prey species in MSVPA:
solid: **both predator and prey assessed within the MSVPA**
horizontal hatching: **external predators eating MSVPA-prey**
vertical hatching: **MSVPA predators eating other food**
cross-hatching: **external predators eating other food.**

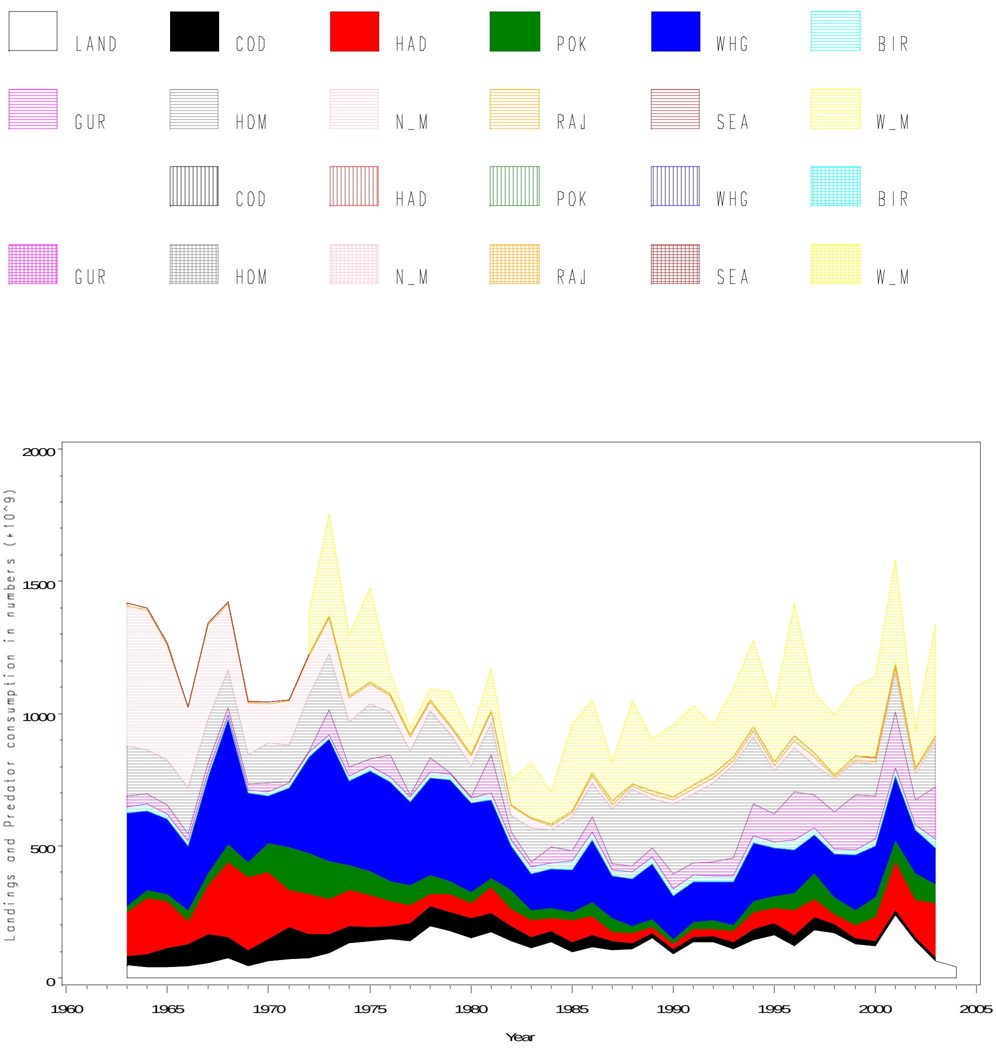


Figure 3.7.2. Time trends in the <u>numbers</u> of fish landed and consumed by predators; results from the key-run. Legends as in Figure 3.7.1. Note that predation on other food (vertical and cross hatching in Figure 4.3 is omitted in this figure. (from ICES 2005d – SG Multispecies assessment for the North Sea)

Combining these three indicators, each individually imperfect, do indicate that fishing has reduced the total abundance of fish in the North Sea. This reduction is seen much more clearly in some parts of the fish community than others, and most strongly in large predatory fish. Because the effect is not general for all types of fish in the North Sea (particularly questionable for some smaller species and ages whose abundance is estimated better by MSVPA than by the surveys) it should be picked up in some of the indices below as well.

3.7.1.2 Richness

Of the two classic components of biodiversity indices, richness (the number of species recorded as present) and evenness (the differential abundance among species present in a community) WGECO has concluded that richness is the more informative of ecosystem status. Daan (2001) examined trends in richness from surveys in the North Sea, concluding that richness is increasing in the southern North Sea. Based on the greater number of Lusitanian species appearing in the surveys, it was concluded that the increasing richness was more likely due to warmer conditions in the southern North Sea than to effects of fishing. In the north-western North Sea, however, reductions in species richness have been demonstrated (Greenstreet & Hall 1996; Greenstreet et al. 1999), and these have recently been linked to fishing activity (Greenstreet & Rogers in press).

3.7.1.3 Evenness

WGECO concluded that Evenness not a reliable metric of community status and trends, and it is not considered in detail here. Some studies have demonstrated fishing effects on diversity metrics that take evenness into account, and linked the observed changes to fishing (eg Greenstreet et al 1999; Greensteet & Rogers in press).

3.7.2 Ecological Functions

3.7.2.1 Trophic relationships

WGECO has considered a number of different types of indicators to reflect trophic relationships, including ones based on food web models, energy flows, and size-based properties. Although the indicators of the first two types are widely advocated elsewhere, ICES and other evaluations have strongly supported the use of size based indicators as the most likely to be informative about alterations of trophic relationships (Rice 2005, ICES, 2006c, Bianchi et al. 2001, Shin et al. 2005, Piet and Jennings 2005, Daan et al 2005).

Daan et al (2005) show clearly that, based on survey results the abundance of large fish (>40 cm) in the North Sea has decline significantly since 1977, and specially since the early 1980s, although again the magnitude of the difference differs among surveys (their Fig 6). By contrast the survey results also show either an increase or no change in the abundance of fish <30 cm. These changes are reflected in an increase in the slope of the size spectrum for all three surveys, although the increase is only statistically significant in the case of one survey (their fig 4). However the changes are even more marked when the relative abundance of species with different maximum possible size (L_{max}) are calculated (their Fig 7). Individuals from species with a maximum body length <30 cm are becoming much more dominant in the North Sea, and individuals from species that may grow to > 50 cm are becoming infrequent. These relative changes in abundance of large predatory fish and smaller fish necessarily represent a change in trophic relationships in the North Sea. This finding is consistent with Heath’s (2005) conclusions, and is reflected in the changing role of predation to total production that he estimates for the different trophic groups (His fig 8), which shows little change from 1973 to 2000 in surplus zooplankton production, but substantial increases in surplus benthic production. Combined with the MSVPA estimates of declining predation mortality on the large fish (ICES 2005d), whatever role top-down control of the ecosystem used to have in the North Sea, it has been weakened greatly, particularly since the late 1980s. Species with small maximum body size, and (from other information) higher turnover rates and greater vulnerability to large forcing by environmental drivers are becoming more dominant.

Daan et al. (2005) attribute their observed changes in the North Sea fish community to the effects of fishing, building on previous work with these data and similar methods by Gislason and Rice (1998) and Rice and Gislason (1996).

Heath (2005) reports a declining trend in benthivores since the late 1980s, with the biomass reduced by as much as 50%. However Frid et al. (1999b) report that the total predator burden on the benthic community in the North Sea may have been increased by as much as much as 25%.. These two estimates have yet to be reconciled, although they may be due at least in part to Heath’s partition of species into fixed trophic guilds, and Frid et al. (1999b) took account of the quite catholic diets of many marine predators and specifically modelled ontogenetic changes in diet. Either change reflects the changes in composition of the predator community, for which we have shown above that fishing was a major cause. Each change also represents big a change in benthos consumed, and it is reasonable to speculate that it has altered at least some aspects of the food web/tropho-dynamic structure.

3.7.2.2 Habitat functions

ICES has identified no reliable indicators of habitat functions. An EcoQ Element has been proposed for habitat quality and extent in the North Sea (Anon 1999) but WGECO has pointed out in past reports (Rice 2005, ICES 2004) that the EcoQ Element cannot be made operational with the data currently available. Undoubtedly in the days before bottom trawling was extensive in the North Sea, the epibenthic communities were more structurally complex and complex communities were more widespread, and there was probably greater structural complexity of the physical habitat with more features of sizes measured in 10's of cm or more. (ICES, 2000a). There is ample research evidence that such structural habitat complexity provides many functions for fish communities, particularly but not exclusively, juveniles of fish that become demersal predators (Patterson et al., 2005, Kenchington et al. 2005, Bremner et al 2005).

Reduction of these epibenthic communities and structural features of the seabed undoubtedly had effects on ecosystem habitat functions, particularly as they affect fish. However, most of such changes occurred many decades ago, and incremental impacts on habitat functions in the North Sea due to continued fishing over the past few decades are likely to be small, at least on the scale of the North Sea as a whole (ICES 2000a). On spatial scales smaller than the entire North Sea, changes in fisheries management regulations have resulted in changes in the distribution of fishing effort in way that have been documented to impact negatively the epibenthic communities and possible habitat structure on spatial scales of several ICES Statistical Rectangles (Dinmore et al. 2003). However, the removal of fishing effort from the closed areas was only seasonal, so it is unlikely that there compensatory improvements in habitat functionality in the closed areas.

3.7.2.3 Nutrients

The ecosystem of the continental shelf typically receives half the nutrients it requires for primary production from the sediment. These nutrients are derived from the dead organic materials which accumulate on the sea floor and which are re-mineralised by bacteria and released back up to the water column by molecular diffusion and biological irrigation. The dynamics of benthic communities are controlled by the depth and overlying production of the water column. Nutrient enrichment of the water column may affect the benthic environment indirectly through stimulation of sedimentary oxygen uptake, rates of ammonification and the release of ammonium from the sediment surface. Some of these benthic communities perform important ecological functions such as irrigating the sediment which strongly affects the nutrient efflux from sediments.

Although there are data on the biota which contribute to nutrient recycling, and some measure of their processing rates, there are very few studies which have measured the direct effects of human activities on nutrient recycling (Percival *et al.*, 2005; Trimmer *et al.*, 2005). The effect of long-term changes in the benthos (see section 3.4.2.5) on nutrient recycling processes are unknown.

Trimmer et al., (2005) showed that fishing had no impact on oxygen uptake, denitrification or nutrient exchange in the southern North Sea. In the long-term, biogeochemical processes in the upper layers of sediment, both oxic and suboxic, appeared unaffected by trawling. This may be because any changes in nutrient recycling which are likely to have occurred as a result of fishing had already happened.

Obtaining direct estimates of nutrient fluxes between the sediment and water column are extremely difficult to obtain and while these would form an obvious metric fro this ecological function this is not practical.

3.7.2.4 Spatial Integrity

ICES concluded that there not even any ways, at present, to set operational ecosystem objectives for “spatial integrity, let alone develop indicators for tracking status relative to such indicators.

3.7.3 Overview of our consideration of community level metrics

Fishing has clearly affected the overall abundance of the fish community in the North Sea, but the effects are size specific. The large-sized component of the fish community has been reduced greatly, whereas the smaller-sized component of the demersal community has not been reduced and may have even increased. This increase may be difficult to reverse, at least as quickly as it was caused, were the current directional pressures of fishing to be removed from the larger sizes of fish, because the fish with small L_{max} have now come to dominate the North Sea fish community. As a consequence of the changes in the size composition and overall abundance of the North Sea fish community, trophic relationships and functions have also changed substantially. Habitat functionality may have changed as will, but at least in the time interval covered by our other data sources it is unlikely to have changed by as nearly as much trophic functionality.

3.8 Conclusions

WGECO solicited this ToR for two reasons, firstly to begin preparations for the North Sea QSR and a likely request from OSPAR and secondly in order to fully test our preparedness to offer advice on the scale of the RAC (see also Section 7). The advantage of using the North Sea as a case study for the latter is that it is, probably, the best-studied sea in the world. Having carried out this assessment to the best of our ability within the time and logistical constraints imposed by the meeting there are a number of relevant questions we now feel able to address. These are:

- How easy was this to do for the North Sea?
- What should be done different in a future exercise?
- Would this be useful for each RAC?
- Should it be repeated on a 6 year cycle?

3.8.1 How easy was this to do for the North Sea?

Describing the anthropogenic activities that affect various components of marine ecosystems is a vital aspect of any “ecosystem approach to management”. Our biggest challenge was lack of readily available data. There were data that we knew existed but which were not available and there were data sets whose reliability was woefully inadequate for the task. Dealing with the issues of data availability should be an immediate priority.

Effort data should be routinely compiled at “international level”, at least by statistical rectangle, and treated with the same level of importance that market sampling and recording of landings are currently accorded. Furthermore, these effort data need to be made freely available to scientists if ecosystem advice is to have a sound basis.

In addition to data availability issues, compilation of these international fishing effort databases has been fraught with problems of data compatibility and quality. For the skippers involved in fishing, the noting of hours actually fished in any day or ICES rectangle is not a “compulsory” field. This has caused difficulties. For the Dutch fleets, for example, recording of effort as hours fishing by vessel per rectangle per trip, if reported by the fishermen, does not appear to have noted in the national database. Consequently, The Netherlands effort data has always been supplied as days absence, and their hour fishing per rectangle per year data effectively “modelled” based on knowledge of the rectangles where landings have been reported, and the fishing characteristics (eg proportion of each day spent fishing, the gear being deployed, etc) of the vessels involved (e.g. Piet *et al.* 2000; submitted). Other countries have submitted actual hours fishing by gear, rectangle and year. For the “Biodiversity” project Scotland were in the latter category. However, in the late 1990s serious concerns regarding these hours fishing data were expressed by FRS scientists, and a detailed analysis of the data collected for 200,000 fishing voyages that fished at least one ICES rectangle in the North Sea over the period 1997to 2004 were examined. This analysis revealed the “hours fishing” data to be seriously flawed and for the “MAFCONS” project, Scotland followed the Dutch lead and supplied “modelled” effort data based on days absence, the rectangles fished (reportedly), and the fishing characteristics of the métiers involved. There is no reason to believe that the Scottish experience is unique, and therefore the reliability of all reported and recorded effort data should be examined. The sentiments expressed in the paragraph above are equally valid in this respect. If an ecosystem approach to fisheries management is to be properly founded in sound advice, then the advice provided needs sound science to underpin it and this can only be achieved if the data available for scientific analysis are reliable.

3.8.2 What should be done different in a future exercise?

Clearly if this were a request from an external customer, OSPAR for the QSR or an RAC, other WGs within ICES would no doubt be able to support the work through the provision of suitable data. This however will require careful planning to ensure requests are directed to the correct source and excessive, unnecessary, requests are not made to already heavily overloaded groups.

In Section 7 we consider ICES readiness to provide advice to the 7 RACS and lessons learnt from our work here with the North Sea are incorporated there.

3.8.3 Would this be useful for each RAC?

As discussed more fully in Section 7, WGECO believe this would be a useful summary document for RACs and of use to them in their work.

3.8.4 Should it be repeated on a 7 year cycle?

It would be feasible to consider this task for 1 RAC per year and by making this an annual exercise develop a 7 year cycle of reporting for each RAC. This could then compliment any additional requests for specific information made in the inter-regum.

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3.10 Technical Annex I: Modelling the direct effects of fisheries on benthos

This technical annex describes a model to predict the spatial distribution of the direct effects of fishing. Models of this form will be an important tool in advising on the consequences of different applications of spatial effort management including MPAs.

Like all models this one makes a number of assumptions, these fall into two groups. Assumptions made in order to make the model tractable and secondly those made to overcome deficiencies in knowledge. Some assumptions may fall into both categories. In this model the fishing effort for each gear type (Section 3.4) is assumed to be distributed evenly within each 1x1 nautical mile (approximately) grid square within a rectangle but the distribution of effort between grid cells matches the observed micro-distribution of effort (described by a Poisson distribution). The impact of a gear on the benthos of an area of sea floor is set, i.e. a set proportion of the benthos is killed. Subsequent, passages of a gear over that area remove *the same* proportion of the remaining fauna. With multiple passages of gear in an intensively fished area, cumulative mortality therefore approaches an asymptote. In reality the proportion of the remaining fauna killed is less on a subsequent gear passes, the most vulnerable individuals are likely to be removed first and subsequent passes occur on a system with a higher proportion of resistant individuals (deeply buried, sheltered by a stone etc.). Because of this the model would tend to OVER ESTIMATE the degree of mortality of the benthos. Neither does the model take any account of the mobility of benthic invertebrates; in its current form it essentially assumes that these are static. Thus no account is taken of the possibility that between fishing events, animals might move tend to move from areas of low fishing activity to areas of high activity simply on the basis of passive diffusion processes. By ignoring these processes, the model would tend to UNDER ESTIMATE the degree of mortality of the benthos. Future development of the model will take account of both these sets of circumstances. For our current purposes, however, these are not issues if the model is used to make comparisons of relative impact, but care should be exercised in considering “exact” levels of impact.

As heavily fished areas reach the asymptotic value of benthic mortality when the proportion of the benthos killed parameter is increased so the asymptote is reached for a lower level of fishing. Thus the model output is essentially a rescaling of the effort distribution maps with the upper ‘bin’ increasing as the level of mortality increases. This formulation is particularly useful in demonstrating that a blanket reduction in effort across the fishery will not deliver the equivalent level of reduction in benthic direct mortality and that more explicit spatial management is required.

If a constant proportion of animals is killed by each passage of a fishing gear, such that the second fishing kills a fixed proportion of animals surviving the first fishing, the third fishing kills the same fixed proportion of animals surviving the second, and so on, then the actual number of animals killed in each subsequent fishing event constantly reduces. The inevitable consequence of this is that, if fishing is not evenly distributed across ICES rectangles such that some patches are fished more frequently than others, the actual mortality caused by fishing (the real “ecological impact of fishing”) will not scale linearly with measures of fishing activity, such as the maps provided in the previous section. For example, consider an ICES rectangle where 20% of the area is fished 5 times and 80% is un-fished. The whole rectangle is therefore fished once on average. If an even distribution of fishing is assumed, and considering an organism with a “per fishing event” mortality rate of 20%, the total number of animals that one might expect to have been killed is 20% of the initial population. In fact 80% of the individuals in the population will have not been aware of the fishing activity going on nearby and all individuals in this un-fished region might be expected to survive. In the 20% of the area fished, only 0.8^2, or 32.8% of the individuals originally present will have survived (and not 0%). Instead of 20% of all the animals in the rectangle being killed, total mortality will in fact only amount to 13.4%.

Recent studies have shown that fishing activity is indeed not evenly distributed across ICES rectangles. Instead, when considered at sufficiently small spatial scale, the distribution of fishing activity follows a Poisson distribution. Thus, when the distribution of both automatic logger position registrations (APR) and vessel monitoring by satellite (VMS) locations across 900 sub-divisions of ICES statistical rectangles (hereafter referred to as sub-units) was examined, the mean:variance ratio tended towards one for all levels of fishing activity within an ICES rectangle (Rijnsdorp & Buys, 1998; Piet et al. 2000). Provided information is available concerning the effect of "individual fishing events" on the benthic organisms present, such as provided by recent meta-analysis studies that have examined the effects of a variety of different fishing gears on different benthic invertebrate species in various habitats (Collie et al. 2000; Kaiser et al., in press), knowledge that the micro-scale distribution of fishing activity follows particular statistical distributions allows much more precise estimates of the impact of fishing within ICES rectangles to be determined (eg. Piet et al. 2000; Piet et al. submitted). The non-linear relationship between measures of fishing activity and the actual ecological impact of fishing can be determined. Here we develop a "generic model" that utilises information about specific fishing activities at the ICES rectangle scale and uses the Poisson distribution to distribute the activity at the micro-scale level within rectangles. Appropriate "community level" mortality rates are assessed based on knowledge of the organisms present in benthic community in different regions of the North Sea and their "per event" mortality rates.

The Poisson distribution determines the probability of observing a specific number of "events" in a particular "cell", given the mean number of "events" across all "cells". Since it deals with "events", the Poisson is an "integer" distribution. When applying the distribution to fishing activity therefore, fishing events must be considered. The micro-scale studies of the Dutch beam trawl fleet considered the distribution of APRs, thus each registration was considered to be an "event" (Rijnsdorp &Buys, 1998; Piet et al. 2000). However, since benthic invertebrate mortality estimates have been determined per fishing trawl, we consider individual trawl tows to be the "events". This also makes sense since the registrations obtained from each individual trawl are certainly not independent of each other. In an ideal world we might have wished to apply the Poisson Distribution directly to the estimates of "Fishing Frequency" per unit space estimated from the fishing activity statistics, since it is these frequencies of event impact that directly drive the estimates of mortality. However, "Fishing Frequency" estimates, ranging as they do from zero to as much as 50 or more as a continuous "real" variable, are not integral in nature, and are therefore not appropriately modelled by a Poisson process (one cannot calculate 2.46 factorial).

The Poisson Distribution is described by the following equation:

$$P(N_{SU}) = \frac{e^{-\bar{x}}.\bar{x}^{N_{SU}}}{N_{SU}!} \quad 1.$$

where $P(N_{SU})$ is the probability of a ICES rectangle sub-unit containing N_{SU} tows when the mean number of tows per sub-unit across all 900 sub-units in the ICES rectangle is $\bar{x}$. To calculate these probabilities for each of the 900 sub-units in any specific ICES rectangle, it is first necessary to estimate the mean number of tows across all 900 sub-units in the rectangle. This is simply done by:

$$\bar{x} = \frac{T_{Rect} / T_{Tow}}{900} \qquad 2.$$

where T_{Rect} is the total number of hours fishing recorded in the ICES rectangle and T_{Tow} is the average tow duration. Substituting equation 2 into equation 1, the probability of any given number of tows occurring in a rectangle sub-unit, from zero to *max* where *max* is the maximum number of tows possible for any particular mean number of tows ($\bar{x}$), can be determined:

$$P(N_{SU}) = \frac{e^{-\left(\frac{T_{Rect}/T_{Tow}}{900}\right)} \cdot \left(\frac{T_{Rect}/T_{Tow}}{900}\right)^{N_{SU}}}{N_{SU}!} \qquad 3.$$

The number of sub-units with all possible numbers of tows can be calculated by multiplying these individual probabilities by 900, the number of sub-units in each ICES rectangle.

To estimate mortality, the "Frequency of Fishing", the number times on average that the whole area in the sub-unit has been fished (FF_{SU}), for each of the rectangle sub-units, needs first to be calculated. This is given by:

$$FF_{SU} = A_F / A_{SU} \qquad 4.$$

where A_F is the total area fished in a rectangle sub-unit and A_{SU} is the total area of the rectangle sub-unit. The area fished is calculated by:

$$A_F = N_{SU} * T_{Tow} * V_{Tow} * W_G \qquad 5.$$

where V_{Tow} is the trawling velocity and W_G is the effective width of the gear. ICES rectangles are 0.5° latitude in height (30NM [x 1.853 = 55.59km]) and 1° longitude in width. While rectangle height remains constant throughout the North Sea, rectangle width decreases with increasing latitude, with consequent decrease in rectangle area. The width of each ICES rectangle is calculated by 60 (minutes longitude) multiplied by 1.853, the conversion factor between NM and km, multiplied by the latitudinal correction factor, the cosine of the latitude in degrees of the ICES rectangle mid-point, Lat_{rect}. Thus the area of a rectangle sub-unit in any given ICES rectangle is given by:

$$A_{SU} = \frac{30*60*1.853^2*\cos(Lat_{rect})}{900} = 6.867218*\cos(Lat_{rect}) \qquad 6.$$

Substituting equations 5 and 6 into equation 4 gives the final equation for estimating the "Fishing Frequencies" in ICES rectangle sub-units in which given numbers of trawl tows have occurred:

$$FF_{SU} = \frac{N_{SU}*T_{Tow}*V_{Tow}*W_G}{6.867218*\cos(Lat_{rect})} \qquad 7.$$

Knowing the frequency that each ICES rectangle sub-unit has been fished on average, and with information regarding mortality per fishing tow (e.g. Collie et al. 2000; Kaiser et al. in press), the total mortality arising from all fishing in the rectangle sub-unit (M_{TOTAL}) can be determined. First the proportion of animals dying per fishing tow (M_{Tow}) must be converted to an instantaneous mortality rate, which can then be multiplied by the sub-unit fishing frequency (FF_{SU}). The result is then converted back to the total proportion of animals dying, thus:

$$M_{TOTAL} = 1 - e^{-Ln[1-M_{Tow}]*FF_{SU}} \qquad 8.$$

Total mortality at the ICES rectangle scale (M_{RECT}) is the average of the mortalities in each rectangle sub-unit. For a given number of hours fishing and with the individual fishing tows distributed across the rectangle following a Poisson distribution at the 0.5° by 1.0° longitude micro-scale, this is calculated by substituting equations 3 and 7 into equation 8, summing over all 900 sub-units and dividing by 900:

$$M_{RECT} = \frac{\sum_{SU=1}^{900} 1 - e^{-Ln[1-M_{Tow}]*\frac{\left(\frac{e^{-\left(\frac{T_{Rect}/T_{Tow}}{900}\right)}*\left(\frac{T_{Rect}/T_{Tow}}{900}\right)^{N_{SU}}}{N_{SU}!}\right)*T_{Tow}*V_{Tow}*W_G}{6.867218*\cos(Lat_{rect})}}}{900} \qquad 9.$$

The non-linear relationship between the measure of fishing activity ($hrs.yr^{-1}.rect^{-1}$) derived from this model is demonstrated in Figure 3.10.1. using beam trawl fleet parameters, where tow duration (T_{Tow}) is 2h, tow velocity (V_{Tow}) is 6.1Kts, and beam trawl width (W_G) is 0.024Km. Total ICES rectangle area, and thus the area of each of the 900 sub-units, assumes a rectangle with a mid-point latitude of 54.75ºN.

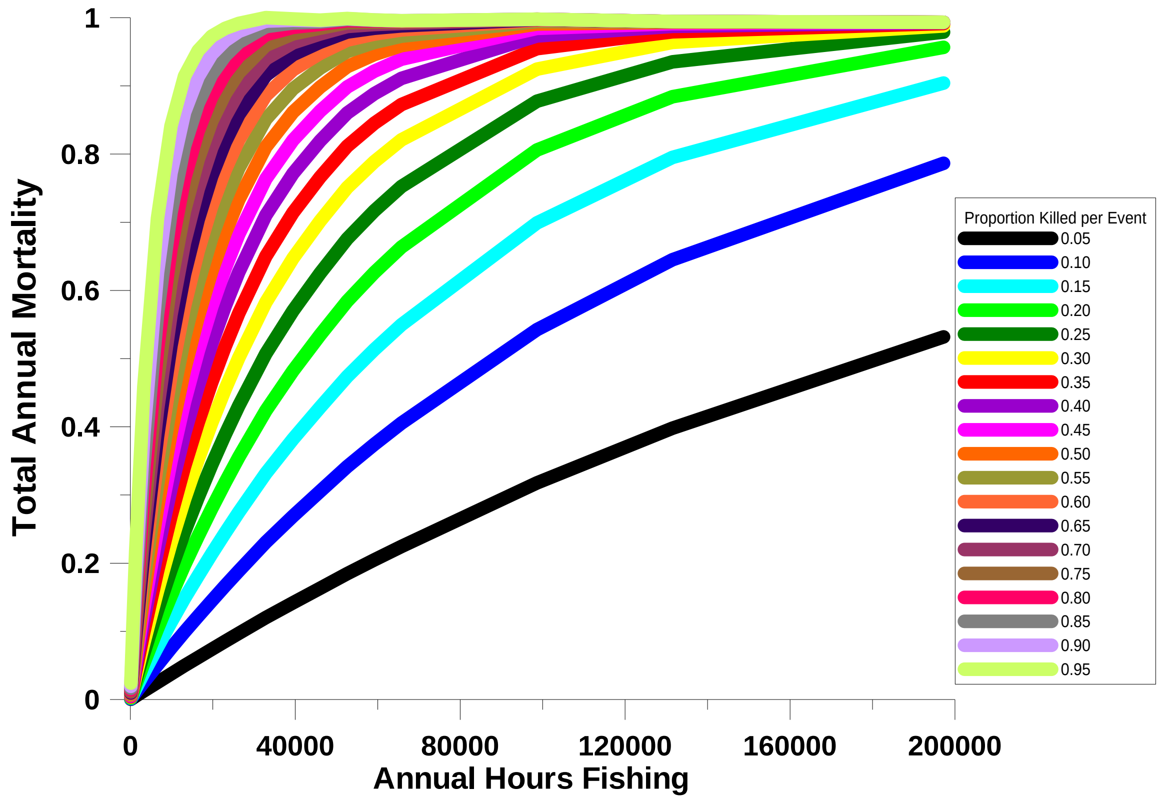

Figure 3.10.1. Relationship between total beam trawl fishing effort in an ICES rectangle and the resultant total mortality of resident benthic invertebrates at various "per fishing event" mortality rates.

3.10.1 Modelling the mortality caused by beam trawling

The model was run using the annual average hours-fishing per ICES rectangle over the period 1998 to 2002 shown in Figure 3.4.1.2, and using published beam trawl fleet parameters of tow duration (T_{Tow}) equal to 2h, tow velocity (V_{Tow}) equal to 6.1Kts, and beam trawl width (W_G) equal to 0.024Km (Rijnsdorp & Buys, 1998; Piet et al. 2000; Piet et al. submitted). Initial runs assumed "per fishing event" mortality rates of 20%, 30%, 40% and 50%. (Figure 3.10.1.1).

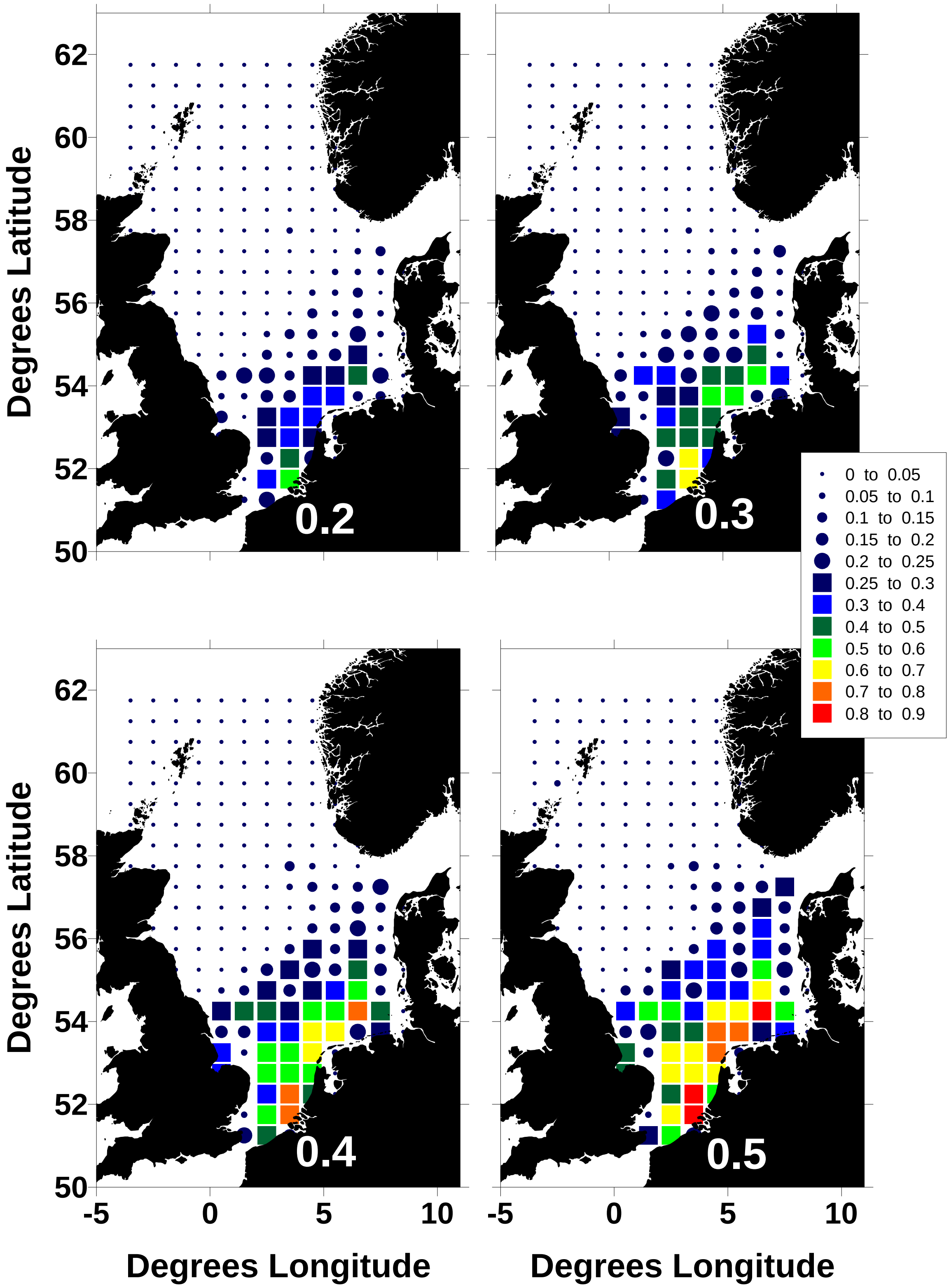

Figure 3.10.1.1. Modelled impact of beam trawling on benthic invertebrates. Maps show total modelled annual mortality given the distribution of beam trawl fishing activity and "per fishing event" mortality rates of 20%, 30%, 40%, and 50%.

3.10.2 Modelling the mortality caused by otter trawling (for fish and Nephrops)

We first examine otter trawling directed at fish. The model was run using the annual average hours-fishing per ICES rectangle over the period 1998 to 2002 shown in Figure 3.4.2.2. Otter trawl fleet parameter data were obtained from published data (Kynoch 1997; Kynoch & Penny 2006), or from unpublished information recorder by observers placed on fishing vessels as part of the discards monitoring scheme. The parameter values used were tow duration (T_{Tow}) equal to 4.7h, tow velocity (V_{Tow}) equal to 2.7Kts, and door spread width (W_G) equal to 0.087Km. Initial runs assumed "per fishing event" mortality rates of 20%, 30%, 40% and 50%. (Figure 3.10.2.1).

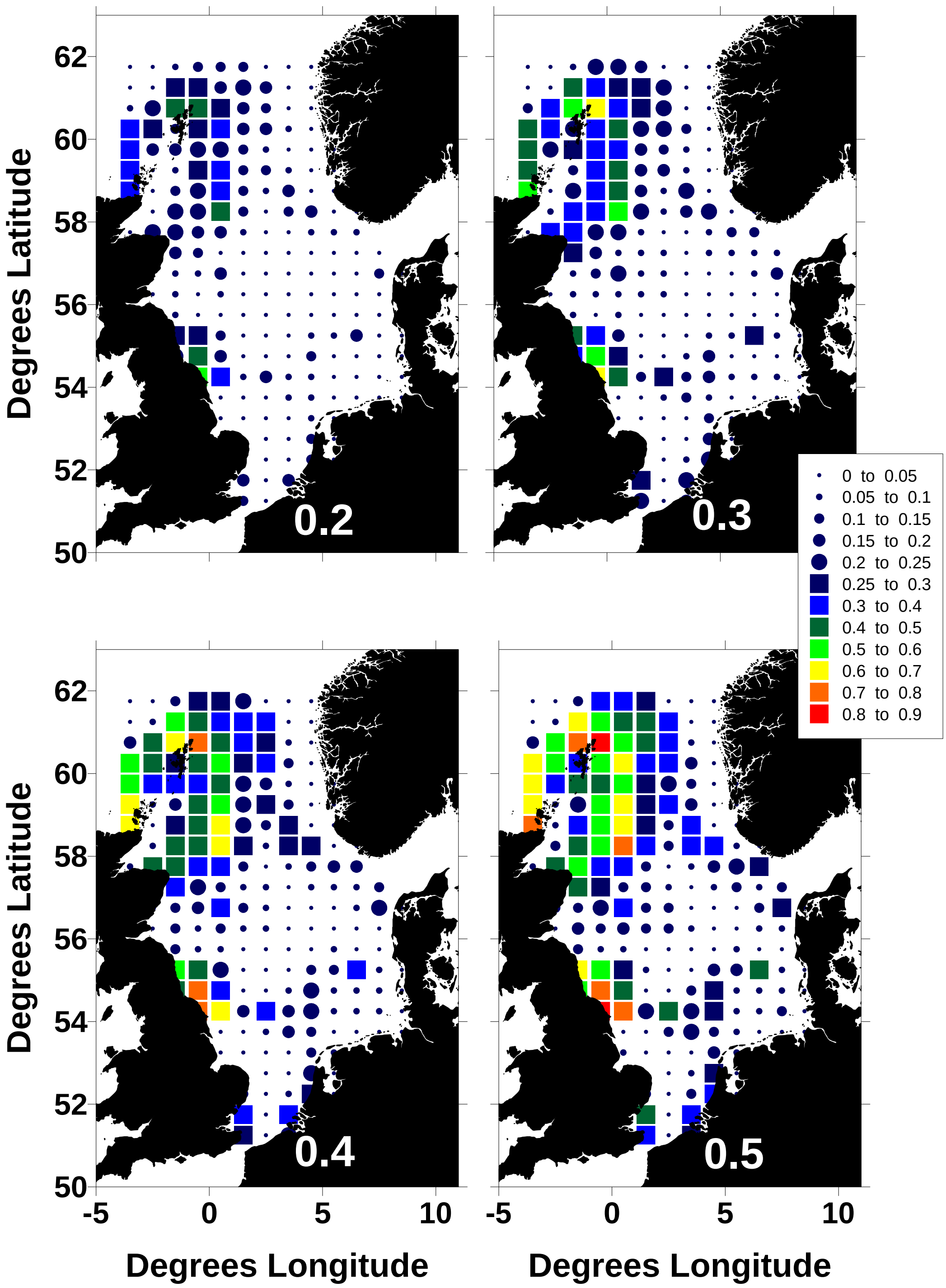

Figure 3.10.2.1. Modelled impact of otter trawling targeting fish on benthic invertebrates. Maps show total modelled annual mortality given the distribution of otter trawl (targeting fish) fishing activity and "per fishing event" mortality rates of 20%, 30%, 40%, and 50%.

We now examine otter trawling directed at *Nephrops*. The model was run using the annual average hours-fishing per ICES rectangle over the period 1998 to 2002 shown in Figure 3.4.2.3. Otter trawl fleet parameter data were obtained from published data (Kynoch 2005), or from unpublished information recorder by observers placed on fishing vessels as part of the discards monitoring scheme. The parameter values used were tow duration (T_{Tow}) equal to 4.9h, tow velocity (V_{Tow}) equal to 2.4Kts, and door spread width (W_G) equal to 0.083Km.

Initial runs assumed "per fishing event" mortality rates of 20%, 30%, 40% and 50%. (Figure 3.10.2.2).

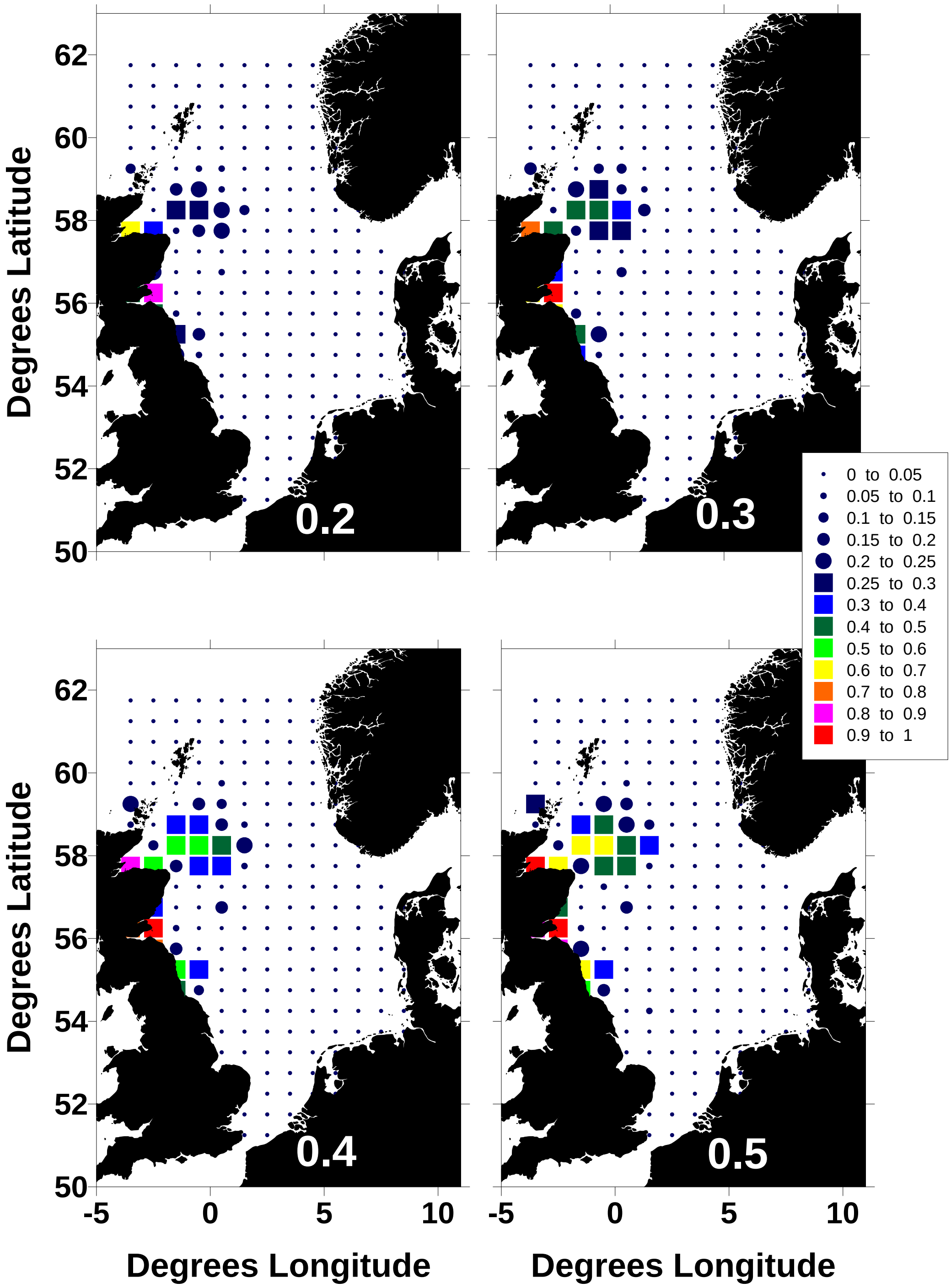

Figure 3.10.2.2. Modelled impact of otter trawling targeting Nephrops on benthic invertebrates. Maps show total modelled annual mortality given the distribution of otter trawl (targeting Nephrops) fishing activity and "per fishing event" mortality rates of 20%, 30%, 40%, and 50%.

3.10.3 Modelling the mortality caused by Seine gears

We now determine the impact of Seine gears on the benthos. The model was run using the annual average hours-fishing per ICES rectangle over the period 1998 to 2002 shown in Figure 3.3.2.4. Analysis of information recorded by observers placed on fishing vessels as part of the discards monitoring scheme suggested that Seine gear tows took on average 1.6 hours once the initial Dan had been picked up. Analysis of the data published by Galbraith and Kynoch (1990) suggested that the average area swept by Seine gear tows of on average 1.6h duration was 2.43km^2. Initial runs assumed “per fishing event” mortality rates of 20%, 30%, 40% and 50% (Figure 3.10.3.1).

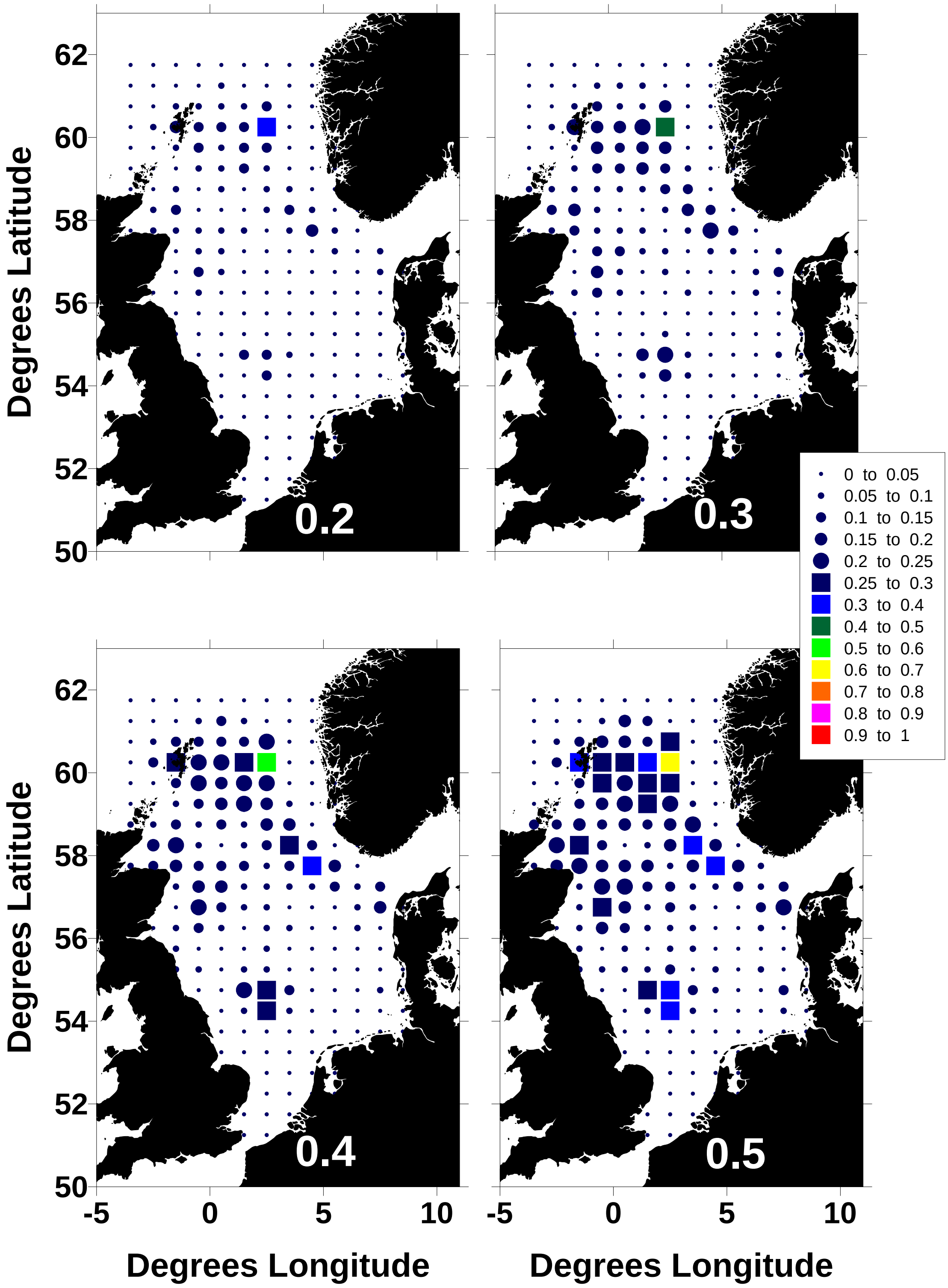

Figure 3.10.3.1. Modelled impact of Seine gears on benthic invertebrates. Maps show total modelled annual mortality given the distribution of Seine gear fishing activity and "per fishing event" mortality rates of 20%, 30%, 40%, and 50%.

3.10.4 References

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3.11 Technical Annex II: Modelling the direct effects of fisheries on fish

This technical annex describes a model that calculates the direct effects of fishing on the fish community, consisting both of commercial and non-target species. Models of this form will be an important tool in advising on the consequences of different types of management measures as part of an ecosystem approach to fisheries management including spatial effort management (e.g. MPAs) or technical measures (e.g. mesh-size regulations). Results are shown in Section 3.5.5

When it comes to assessing the abundance and removal of fish from the ecosystem by fisheries it is useful to distinguish between commercial species and non-target species. These two components not only differ in that certain fisheries specifically target the commercial species but also with regard to the availability of knowledge and data on the effects of fishing.

Fishing gears catch individuals of both commercial and non-target species (Heessen and Daan 1996). What is retained in the net is determined by characteristics of the fish and gear selectivity. The part of the catch comprised of non-target species and damaged, undersized and juveniles of target species is considered by-catch. Some of the captured non-target species are of economic importance and will be landed, whilst other species, which have no economic importance, are discarded. Discards may also include damaged, undersized and juveniles of target species.

The extensive sets of data that exist for the target species and can be used with (Multi-Species) Virtual Population Analysis, (MS)VPA, to estimate stock abundance and various sources of mortality, do not exist for the non-target species. Yet these species not only make up a significant part of the biomass but also fulfil an important role in the functioning of the ecosystem and are often of interest from a conservation perspective (e.g. sensitive species like most elasmobranchs). Therefore we developed a method that allows estimation of abundance and the impact of fishing (i.e. bottom trawling) on these species in terms of the proportion removed or mortality. This type of information can be used together with information on the life-history characteristics of species to determine if levels of exploitation are sustainable.

The fish absolute abundance estimates are based on (MS)VPA and only exist for the main commercial species. These estimates, together with catch data from extensive monitoring programs that provide (relative) estimates of abundance of the main non-target fish species were used to obtain absolute abundance estimates of the non-target fish species (Yang 1982, Sparholt 1990). This approach was based on the assumption of equal catchability of non-target and commercial species with similar characteristics. In this approach, however, the size component was ignored.

Fishing mortality estimates for the North Sea are available for 10 commercial fish species that are assessed routinely to provide annual advice on Total Allowable Catches (TACs) (ICES 2002, 2003). The principal method used is (Multi-Species) Virtual Population Analysis (MSVPA), which requires reliable estimates of the age composition of the total international catches and allows an evaluation of the historic development of fishing mortality and stock numbers by age group up to the present day. While this method yields converged parameter estimates for year classes that have reached the end of their life, estimates for recent years will vary to some extent during subsequent assessments, because of uncertainty about the proportion still surviving. To obtain the best possible estimates, a variety of statistical methods has been developed that use additional information on catch per unit of effort (CPUE) derived from commercial and/or research vessel data to tune initial estimates of fishing mortality (Shepherd 1999).

Two potential methods to estimate mortality rates are (1) an extension of Jones's (1981) length cohort analysis and (2) an approach based on estimates of swept areas from fishing fleets (Pope *et al.*, 2000). The latter method may be easier to use as, unlike the length cohort analysis method, the swept area method does not require sampling of commercial by-catches and the required data on the distribution and abundance of non-target species are often available from surveys (Kunitzer *et al.*, 1992; Knijn *et al.*, 1993).

The direct effects of fishing expressed as the mortality/removal of a specific ecosystem component in a spatially disaggregated system is determined by:

- the abundance of that component in each spatial unit
- the frequency with which each spatial unit is fished with a specific type of gear
- the impact of the singular passing of the gear expressed as the proportion of that component removed
- the integration of the above over all spatial units

This approach is used in a spatially disaggregated Direct Effects Model (DEM) which we use to estimate the size- and species-specific mortality caused by bottom trawling.

Absolute abundance

To estimate the abundance of non-target species we applied a slightly modified version of the method developed by (Sparholt 1990). We followed (Sparholt 1990) in that we estimated the abundance of non-target species by combining MSVPA-based abundance estimates of target species with survey catches that include both target and non-target species but improved the method in that we retained a size component and the spatial structure inherent to the survey data. For this Sparholt distinguished the following groups: (1) cod , haddock, whiting , saithe; (2) Norway pout; (3) herring, sprat; (4) sandeel; (5) mackerel; (6) plaice; (7) sole.

VPA and MSVPA are assumed to provide the most accurate estimates of stock abundance at age in the North Sea of a suite of commercial species. Both approaches provide comparable abundance estimates (Table 3.11.1) but an advantage of MSVPA is that all abundance estimates are standardized to one area (i.e. ICES area IV). Therefore this was used as the best estimate of stock abundance.

Table 3.11.1. Mean ratio of MSVPA/VPA abundance estimates for the main commercial species-at-age, period 1985-2000. The VPA abundance estimates were not corrected for the size of the area.

Species		Age			
Scientific name	Common name	1	2	3	4+
Ammodytes sp.	Sandeel	0.89	1.13	1.23	1.38
Clupea harengus	Herring	0.60	1.15	1.16	1.09
Gadus morhua	Cod	0.49	0.65	0.80	0.83
Melanogrammus aeglefinus	Haddock	0.66	0.88	0.96	1.06
Merlangius merlangus	Whiting	1.39	1.00	1.07	1.17
Pleuronectes platessa	Plaice	0.36	0.44	0.70	1.04
Pollachius virens	Saithe	0.90	0.90	0.93	0.89
Solea vulgaris	Sole	1.01	1.03	1.03	1.05
Sprattus sprattus	Sprat	1.42	0.58		
Trisopterus esmarki	Norway pout	1.16	0.92	1.02	

The catch rates of most commercial species-at-age in the North Sea are based on two surveys (Figure 3.11.1):

- Beam Trawl Survey (BTS) for Sparholt groups 6 and 7
- International Bottom Trawl Survey (IBTS) for all other groups

Because the MSVPA estimates of stock abundance are all fixed on the first day of the year the assumption is that this date is best represented by the 1st quarter IBTS catches. As the BTS takes place in the 3rd quarter the MSVPA abundance in the 3rd quarter ($A_{3,a,y}$) were derived from those in the 1st quarter according to:

$A_{3,a,y} = (A_{1,a,y} + A_{1,(a+1),(y+1)})/2$

We used catchability to convert the survey catches into absolute abundances. Catchability is calculated as IBTS catch-rate ($N.km^{-2}$) divided by MSVPA abundance ($N.km^{-2}$). For the commercial species catchability was first determined per age group. Then the commercial species were divided into 5-cm length groups up to sixty centimeter above which catchability was assumed constant and they were grouped together (see Table 3.11.2). For each 5 cm length-group the catchability of the age-group that contributes most to this length group was used. The linking of length and age-groups was based on the survey-based age-length keys of each species. Age 0 was not included in the analyses. Although both surveys are not suited to catch fish below 10 cm, all that were caught were grouped together and attributed the catchability of Age 1. To convert the number per haul of each survey to a number per fished area, we assumed an effective width of the IBTS of 15 m and BTS of 8 m. Both surveys conduct half hour tows with a fishing speed of 4 knots (=1852 $m.hr^{-1}$) resulting in a fished area of approximately 0.056 km^2 per tow for IBTS and 0.030 km^2 for BTS.

We followed (Sparholt 1990) and divided the commercial species into the same groups with the exception that mackerel was not included in the analysis and saithe was not considered part of group 1 because only a small part of the saithe stock is found in the North Sea resulting in an underestimation of its catchability. This resulted in the catchabilities displayed in Table 3.11.2.

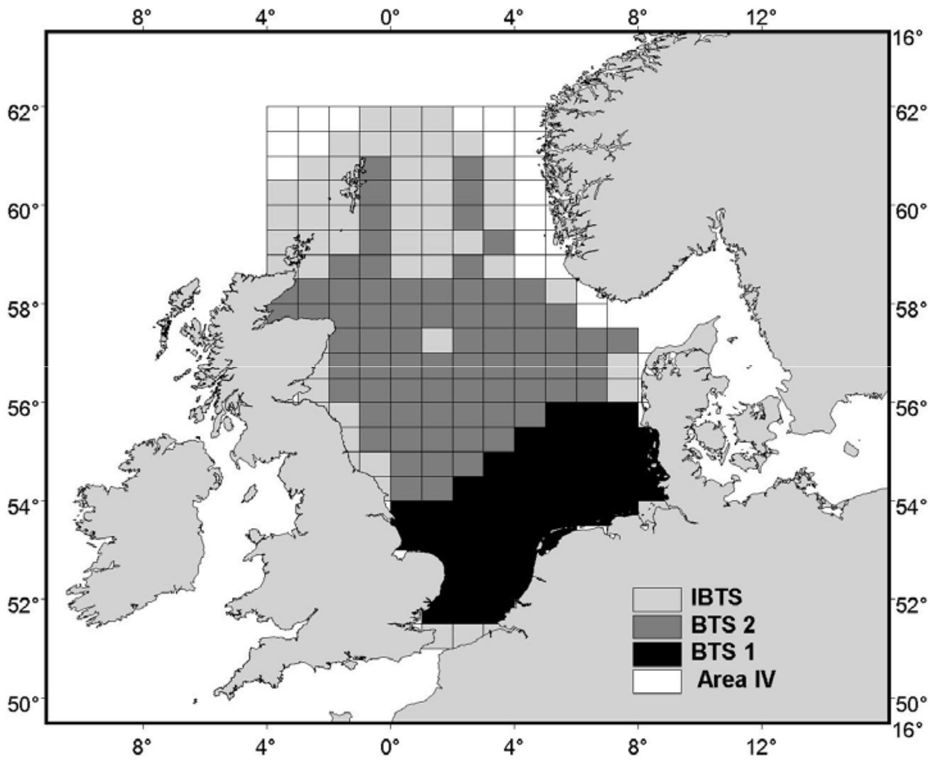

Figure 3.11.1. ICES area IV and the rectangles covered by surveys IBTS and BTS (BTS 1: 1985-1995, BTS 2 1996-2000). All areas covered by a darker shade are also covered by the lighter shaded surveys.

Table 3.11.2. Catchabilities of the commercial species per length group (as mean values for 1985-2000) based on survey and MSVPA data. For the species based on IBTS the effective width of the gear was based on the wingspan.

Species	Spar holt	10	15	20	25	30	35	40	50	60
Ammodytes spp. (x1000)	4		0.81	1.03	0.90	0.94				
Clupea harengus	3	0.28	0.30	0.14	0.12	0.12	0.12			
Gadus morhua	1	0.16	0.31	0.48	0.57	0.70	1.07	1.16	1.22	1.20
Melanogrammus aeglefinus	1	0.59	0.94	1.65	1.42	1.20	1.18	1.18	1.18	1.18
Merlangius merlangus	1	0.65	1.58	1.39	1.17	1.16	1.16	1.16	1.16	1.16
Pleuronectes platessa	6	1.69	1.74	1.67	1.16	1.00	0.43	0.39	0.39	0.39
Pollachius virens			0.01	0.01	0.01	0.03	0.06	0.11	0.11	0.12
Solea vulgaris	7	0.46	0.42	0.42	0.50	0.40	0.41	0.47	0.48	0.43
Sprattus sprattus	3	0.12	0.20	0.20						
Trisopterus esmarki	2	0.31	0.68		0.38					

For the Sparholt groups consisting of several species (e.g. groups 1 and 3) the catchabilities were calculated by taking the un-weighted mean of the species in each group. All other species caught are allocated to one of these six groups, and we assume that a non-target fish of equal size as a commercial fish in a particular group also has an equal catchability. If there is no value for catchability for a particular Sparholt/size-group in a particular year then the 3-point moving average value is used and if that is also missing the mean catchability of that size-group is used until finally the mean catchability of the Sparholt group is used to calculate abundance if all else is missing.

Trawling frequency

The frequency with which an area is trawled is considered to be a better measure of fishing impact than conventional effort measures such as days-at-sea or hours fished. In Table 3.11.3 are quantitative data on relevant fishing parameters (e.g. the proportion of the day actually spent fishing, fishing speed) and gear characteristics (e.g. width of the gear) that allow the transformation of these conventional measures into trawling frequencies. “Eurocutters” cover on average an area of 1.2 km^2 each day, while “Large vessels” cover an area of 5.3 km^2.

Table 3.11.3. Fishing parameters of two métiers of the Dutch beam trawl fleet.

Métier	Speed (knots)	Hours fishing per 24h	Proportion of the day spent fishing (%)	Area (km^2) swept per day
Eurocutter	4.2	19.3	80.4	1.2
Large vessels	6.7	17.7	73.9	5.3

For fishing effort we used the data from the MAFCONS project on the international otter- and beam trawl effort for the period 1998-2002. Trawling frequency (F) is calculated as:

$F = Eff_w \times T_F \times S \times S_{ICES}^{-1}$

Where:

F = Frequency trawled

Eff_w =Effective width (m)

T_F =Time Fished (s)

S=Speed (m/s)

S_{ICES}=Surface of ICES rectangle (m^2)

Impact of the gear: catch efficiency

In the model the direct effect of a fishery on a species is determined according to (Pope et al. 2000) but improved with regard to the assumption of a 100% catch efficiency. The interaction between fish and bottom trawls is a complex issue and determined by fish behaviour in relation to gear characteristics, making the catch efficiency of a gear hard to quantify (Wardle, 1988;Dickson, 1993). Based on the available literature (Weinberg et al., 2002;Engås and Godø, 1989) we developed a conceptual framework in which catch efficiency is determined by four factors:

- Positioning in the water column
- Herding
- Escape below footrope
- Retention in the net

Some of these factors are discussed in more detail below. There are numerous other factors that may affect catch efficiency. For example vessel noise (Dickson 1993), visibility, fishing speed, density-dependent catchability, diel variation and mesh shape (Wardle, 1988; Weinberg et al., 2002; Godo et al., 1999; Benoit & Swain, 2003; Robertson et al., 1988). The lack of quantitative data, however, prevented us from incorporating these factors.

The positioning in the water column of the fish relative to the gear determines the likelihood that fish enter the mouth of the net. As there are no quantitative data we assume that 90% of the roundfish are positioned such that they do not succeed in escaping over the headline of the otter trawl and as a beam trawl has a markedly lower vertical opening this is assumed to be only 20% for the beam trawl. Flatfish are assumed not to be able to pass over the top of both types of gear.

Not all fish species between the otter boards are herded towards the mouth of the net (Wardle 1986; Dickson,1993; Engås & Godø, 1989; Ramm & Yongshun, 1995). For roundfish, Engås & Godø (1989) compared the catches of cod and haddock between gears with different sweep lengths. With increasing door-spread, a significant increase was found in catches for cod and haddock, especially for larger fish lengths (Engås and Godø 1989). However, for simplicity herding can be assumed independent of fish length (Ramm & Xiao, 1995). Herding is assumed to be related to difference in door-spread between gears. Engås & Godø (1989) used an average herding effect per meter door-spread of 0.067 and assumed a standard otter trawl has a sweep-length of 40m, a door-spread of 58m, a net opening of 19m, and a difference of 39m in width between door-spread and net opening resulting in a correction factor for the proportion of fish that do not reach the mouth of the net equal to (39∗19∗0.067)/58=0.85. Even though the dimensions that we assumed apply for the commercial otter trawls in the North Sea (Wingspread=26.6 m and Doorspread=86.6 m) are different from those on which the calculation of the herding factor was based we used this factor for the larger (>29 cm) roundfish. No quantitative data on herding were found for flatfish. According to Winger (1999) larger flatfish should be capable of reaching the net opening. However, Winger (1999) assumed a towing speed markedly lower than that of the fishing fleet in the North Sea and as (Wardle 1988) showed that the endurance rapidly decreases with increasing speed we assume a correction factor equal to wingspread/doorspread (i.e. no herding) for flatfish and small (< 29 cm) roundfish.

The proportion of fish passing below the footrope is dependent on species, size, fishing speed and gear construction and reduces the efficiency of the gear (Engås and Godø 1989) (Dahm 2000) (Weinberg et al. 2002). Estimates of the proportion passing below the footrope results in an efficiency of 0.95 for roundfish while for flatfish we used a footrope factor of 0.5 for smaller (< 0.25cm) flatfish and 0.85 for larger (≥ 25cm) flatfish (Weinberg et al. 2002).

Most fish are considered to escape from the cod-end of the gear (Millar and Fryer 1999) and therefore most studies on gear selectivity have been carried out on cod-end selection (Wileman et al. 1996). Gear characteristics such as mesh size, cod-end extension length, cod-end diameter or mesh-shape have a significant influence on the selection of fishing gears (Beek et al., 1981, 1983;Reeves et al., 1992; Robertson et al., 1988; Zuur et al., 2001). The proportion of fish that is retained in a net is calculated as a function of mesh size using cod-end selectivity data. (Wileman, 1991) summarized several gear selectivity studies carried out over a period of more than 30 years. Several species in two types of gear were distinguished: seven species in the otter trawl (OT) and two in the beam trawl (BT) (Table 3.11.2).

A logistic curve is used to describe the relationship between the length of a fish and the proportion of a population that is retained in a net (Casey 1996):

$PR_L=\{(3^{(L50-(L+\Delta L/2))/(L50-L25)})+1\}^{-1}$

Where:

PR_L = The proportion of the population of length L and class width Δ L that is retained.

L50 = The length of which 50 percent of the population entering the net is retained (cm)

L25=The length of which 25 percent of the population entering the net is retained (cm)

L50 and L25 are calculated from the selection factor (SF) and selection range (SR) according to (Wileman 1991; Wileman et al., 1996).

L50 = SF * M

L25 = L50-(SR/2)

Where:

SF= Selection Factor

M= Mesh size (cm)

SR=Selection range (cm)

As sufficient quantitative information to determine cod-end selectivity is only available for some commercial species (Maclennan et al., 1992) we determined selectivity parameters for roundfish and flatfish and applied those to the non-target species.

The values for the positioning, herding and footrope (Small/Large fish) factor are assumed constant (Table 3.11.3). These factors are multiplied to result in a final efficiency factor. Thus a beam trawl is more selective than an otter trawl for flatfish (1 versus 0.12 for small and 0.21 for large flatfish) and less selective for roundfish (0.6 versus 0.77 for roundfish) (Table 3.11.4).

Table 3.11.3. Factors used in the direct effect model for calculation of catch efficiency for beam trawl (BT) and otter trawl (OT) and different fish types, demersal roundfish (DR), demersal flatfish (DF) and pelagics (P). The factor is dependent both on fish-size and mesh-size. The footrope factor is divided in a factor for smaller (S, < 25 cm) and a factor for larger (L, ≥ 25 cm) fish. The overall factor (L/S) is calculated by multiplying the positioning, herding and footrope factor.

		Factor			
Gear	Fish type	Positioning	Herding	Footrope	Overall(L/S)
BT	DR	0.2	1	1	0.20
BT	DF	1	1	1	1
BT	P	0.05	1	1	0.05
OT	DR(L/S)	0.90	0.85/0.3	0.95	0.73/0.26
OT	DF(L/S)	1	0.3	0.7/0.4	0.21/0.12
OT	P	0.25	0.85	0.95	0.20

Table 3.11.4. Gear selectivity parameters selection factor and selection range for different species and species groups. Two types of gear have been used. OT=Otter trawl, BT=Beam trawl. Mean values for roundfish and flatfish species have been calculated. Note that the mean value for flatfish does not include sole.

Species	Geartype	Selection factor	Selection range (cm)
Cod	OT	3.0	7.2
Haddock	OT	3.1	6.6
Whiting	OT	3.5	6.6
Saithe	OT	4.3	5.7
Dab	OT	2.5	1.9
Plaice	OT	3.3	1.6
Sole	OT	3.4	4.1
Dab	BT	2.2	4.1
Plaice	BT	2.2	3.6
Sole	BT	3.2	3.9
Roundfish	OT	3.5	6.5
Flatfish	OT	2.9	1.8
Flatfish	BT	2.2	3.9

On-board selection

On-board selection determines which part of the fish caught, are actually landed, the remainder being discarded. Discards can either be target species that, according to regulations, are too small to be landed or species of no commercial interest. (Casey 1996) suggested a logistic curve to approximate the selection process but as we had no other information than the minimum landing size we used this to estimate discards. If a fish species has no minimum landing-size we assume that the species is completely discarded.

Mortality

The mortality (M) expressed as the number of individuals caught is calculated from the Proportion retained (PR_g) and the Trawling frequency (F_g) by first calculating the chance that an individual is not retained by a specific gear g:

$C_g=(1-PR_g)^{**}F_g$

If two metiers are considered such as in this simulation, the mortality is:

$M = N^*(1-C_b{}^*C_o)$

Where N is the number of individuals present in the path of the gear.

The fish caught can be divided into landed or discarded fish based on their qualification as commercial species or non-target species and in case of the first, the minimum landing-size.

The % discarded is calculated as the number discarded/number caught.

The % mortality is the number caught/ number present

3.11.1 References

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4 TOR b) Assessing the key pressures on marine ecosystems

Complete the identification and selection of key pressures of human activities on the state of the marine ecosystem begun in 2005, and identify indicators, metrics, data series and reference levels (as appropriate) for these pressures.

4.1 Introduction

In the ICES area there are several initiatives to develop Integrated Assessments (IA) for national areas of jurisdiction and/or regional seas such as the North Sea. The Regional Ecosystem Group for the North Sea (REGNS) has moved towards assessing variability in the structural and functional aspects of components that make up the North Sea ecosystem by bringing together most data available and carrying out a number of analyses of state (ICES 2004a, 2005a) (see section 8). Other bodies have made assessments by describing the important impacts and pressures that affect the status of the marine ecosystem, with an interpretation of whether these pressures and their associated impacts are changing based on the best available knowledge ((Defra, 2005; EC, 2005; UNEP, 2005). Similar attempts have been made in Canada (e.g. DFO, 2003; (Choi et al., 2005)) and tools such as the “traffic light” approach (as described by (Choi et al., 2005)) have been used on both sides of the Atlantic. Although these approaches can be useful for summarising trends in key ecosystem properties and threats, and providing an overview of the ecosystem status of regional seas, they do not allow links to be made between manageable human activities and the variability of ecosystem components which is a key requirement of an IA (see ICES, 2005b).

In 2005, WGECO identified that a further step was required to address the development of a full Integrated Assessment; namely, to develop a formal framework to link manageable human activities with the pressures they cause in the marine ecosystem through the actual mechanisms with which pressure is exerted on the ecosystem (e.g. selective extraction of individuals; smothering etc.) (ICES, 2005b). Similar initiatives are now ongoing as part of the developing national and international marine strategies (e.g. EC, 2005). Building on progress made in Canada (e.g. DFO, 2003; Choi *et al*., 2005), a two-table matrix was developed; the first table associates individual ecosystem components with specific pressures and the second links those pressures to the activities which are responsible for them. This framework provides a useful tool for identifying all the pressures of human activities on the state of the marine ecosystem. To be fully effective, this process should convey two types of information in order to provide a mechanism for identifying the priority pressures on each individual component. One type of information is about the strength of the interaction in the ecosystem; how reliably is the pressure-state interaction expected to occur, if the pressure is being applied in the real world. We consider this the “weight” of the pressure-state interaction for each ecosystem component. The other type of information is how serious the interaction is to the ecosystem and/or to society: if it occurs, should management do anything about it. In this ToR we deal with the first of these tasks, but do so in the context of being informative about the second. That is, the way that the “weights” are distributed should inform managers which types of options for management action are most likely to be needed. The first aim of addressing this ToR, therefore, is to complete the development of the IA framework started by WGECO in 2005, by weighting the key pressures for each ecosystem component. Having done so, we explore the metrics, indicators, data series and reference levels that could be used to monitor these key pressures in the North Sea ecosystem (section 4.2).

In 2005, WGECO described the approach that could be applied to utilise this framework in linking change in state of the marine ecosystem with manageable human activities through the pressures that these activities cause. In section 4.3 we describe the metrics available to monitor the key pressures identified within this approach using the fish component as an example. We also describe an alternative approach that would require a similar suite of metrics but would be more useful for regulation of particular activities by sector.

4.2 Key pressures and ecosystem components

4.2.1 Defining the approach

We limit ourselves here to an assessment of the IA framework in a fully marine offshore environment. Although we concentrate on pressures that affect ecosystem components in the offshore ecosystem for significant periods of their life cycle, we recognise that some of these components will spend periods of time in the coastal zone (e.g. seabird nesting sites, marine mammal (seal) haul outs). Thus, although a particular ecosystem component may not be affected by a specified pressure whilst offshore, it may be in the coastal zone, suggesting that a fully comprehensive IA should ultimately take account of both coastal and terrestrial pressures and impacts.

4.2.2 Ecosystem components

The list of ecosystem components considered is based on the WGECO definition (see ToR (a)) (ICES, 2004b) but extended here to reflect differences in susceptibility to particular pressures. For example, ‘physical habitat’ (the structural features of habitat) was separated from ‘water column and bio-chemical habitat’ because pressures such as smothering and substratum loss only apply to the physical habitat, whilst those related to contamination or changes in nutrient levels, for example, only apply to the water column and bio-chemical habitat. The category ‘Fish’ included both non-target fish and target fish species of commercial importance as both are likely to be affected by the same pressures at this level. Finally, we have separated macrophytes from other components of the benthos, identified discrete categories for zooplankton and phytoplankton, included cephalopods and a category for marine reptiles. Comprehensive coverage of the ecosystem in this approach can only be provided by considering the specific components and their attributes, where components are functional or species groups and attributes are their properties.

4.2.3 Pressures

The list of physical, chemical and biological pressures was developed by WGECO (2005b) to provide a broad overview of the effects of human activities in the offshore marine environment. It is based on pressures used for the interpretation of the EU Habitats and Birds Directives, and is broadly comparable to that in the Annexes to the draft EU Marine Strategy Directive (EC, 2005; EU, 1992).

4.2.4 Weighting the significance of interactions between pressures and ecosystem components

At a workshop to identify ecological objectives for the marine environment (Rogers & Tasker, 2005), an exercise was undertaken to identify the strength of links between ecosystem components and pressures caused by human activities (strong, weak, none or unknown). Strength of the interactions was closely related to the spatial scale at which the assessment was undertaken, however, scoring for the intensity of the effect on a component was less transparent. In this ToR we have further developed this approach and have scored the interactions between pressures and ecosystem components based on their spatial extent and their intensity. Spatial extent (local (L) or widespread (W)) was defined relative to the area covered by an individual ecosystem component in the ecosystem being assessed. The intensity of the pressure (chronic (C) or acute (A)) was defined in terms of its’ duration. Chronic interaction was used to describe a pressure that lasts for a long period of time or is marked by frequent recurrence, but where even the cumulative effects may not lead to any or a significant proportion of component level mortality or destruction. It may also include indirect effects to a component (e.g. changes in growth rates brought about by a change in temperature or decreased productivity of the benthos due to reduced productivity from the plankton based on increased turbidity levels). Acute pressure was defined as a relatively short but intense and instantaneous interaction, and causing mortality or destruction to a component at a high proportion of the component or populations included.

In Europe it is expected that integrated assessments of, for example, ecosystem health, are likely to occur at an approximately decadal scale, to match existing management procedures such as the OSPAR QSRs. In recognition of that temporal scale, we have constrained the weighting of interactions to reflect the strength of the pressure within a fixed time period. Thus, although a pressure may lead to a significant change in a population over a multi-decadal timescale (e.g. the biogeography of a population may be reduced to one fifth of its former extent in the North Sea due to sea temperature warming over a 10-20 year period), the approach we describe would score the effects of this pressure as chronic. This is to account for the relatively small adverse change that would occur over a 10-year period at the scale of the entire population.

In all cases weighting is based on a judgement of a plausible ‘worst-case’ scenario for an ecosystem component, recognising that different communities, species or populations may have a weaker interaction. Extent and intensity were both scored based on expert knowledge of the ecosystem of interest and its pressures relative to its recent history (i.e. last 10 years). For the purpose of this exercise we focus on the North Sea as an example.

4.2.5 Results: what are the key pressures?

Widespread and acute (WA) interactions are considered to highlight the most important pressures (key pressures) because they will cause a large and rapid change in an ecosystem component over a significant extent of the area being assessed. In the North Sea ecosystem, the key pressures that result in such interactions are ‘Abrasion/ physical disturbance’ for the benthos, macrophytes and the physical habitat. This is likely to have important consequences for the ecosystem as a whole given that a widespread change in the physical and biological benthic components will alter the likely composition of the rest of the ecosystem. The ‘Selective extraction of species’ is a key pressure for the fish, cephalopods and benthos. ‘Contamination by synthetic compounds’ and ‘Introduction of microbial pathogens/parasites’ are both key pressures for the marine mammals when based on plausible worst-cases of the North Sea in the last 10 years. ‘Introduction of non-native species and genetically modified organisms (GMOs)’ is a key pressure for seabirds in the recent North Sea system (Table 4.2.5.1).

Table 4.2.5.1. The interaction between specific pressures and individual ecosystem components. LC= locally chronic; LA = locally acute; WC = widespread chronic; WA = widespread acute; U=unknown; N=none or negligible effect. To be fully effective for the selection of indicators, the ecosystem components must also include attributes of species, communities and populations. The column 'worst case' identifies the components (using their abbreviated forms given in brackets e.g. Fish = F) for which each pressure has the strongest interaction (strongest WA>LA, A>C, WC>LC weakest).

Ecosystem component		Fish		Benthos		Plankton		Seabirds	Marine mammals		Habitat and Nutrients		
		Fish (F)	Cephalopods (C)	Benthos (B)	Macrophytes (M)	Zooplankton (Z)	Phytoplankton (P)	Seabirds (S)	Marine mammals (MM)	Marine reptiles (MR)	Water column & bio-chemical habitat (WH)	Physical habitat (PH)	Worst-case
Pressure													
Physical	Substratum loss (e.g. by permanent constructions)	LC	LC	LA	LA	LC	LC	LA	LA	LA	N	LA	B, M, S, MM, MR, PH.
	Smothering	LA	LA	LA	LA	LC	LC	N	N	N	LC	LA	F, B, M, C, PH
	Change in suspended sediment	LC	LA	LA	LA	LC	LC	LC	N	N	LC	LA	B, M, C, PH
	Change in water flow rate	LC	LC	LC	LC	LC	LC	N	N	N	LA	LC	WH
	Change in thermal regime (e.g. outfalls, power stations)	LA	LA	LC	LC	LA	LA	LA	N	N	LA	N	F, Z, P, S, C, WH
	Change in temperature (climate change)	WC	WC	WC	WC	WC	WC	WC	WC	WC	WC	N	F, B, Z, P, M, S, MM, MR, C, WH
	Change in turbidity	LC	LC	LC	LA	LC	LA	N	N	N	LA	N	P, M, WH
	Change in sound field	LC	LA	LC	LC	N	N	N	LA	U	N	N	MM
	Change in light regime	LC	LC	LC	LA	LC	LA	N	N	LC	LA	N	P, M, WH
	Visual presence	N	N	N	N	N	N	LC	LC	LC	N	N	S, MM, MR
	Abrasion/ physical disturbance	LC	LA	WA	WA	N	N	N	LC	LA	N	WA	B, M, PH

Table 4.2.5.1.continued. The interaction between specific pressures and individual ecosystem components. LC= locally chronic; LA = locally acute; WC = widespread chronic; WA = widespread acute; U=unknown; N=none or negligible effect. To be fully effective for the selection of indicators, the ecosystem components must also include attributes of species, communities and populations. The column 'worst case' identifies the components (using their abbreviated forms given in brackets e.g. Fish = F) for which each pressure has the strongest interaction (strongest WA>LA, A>C, WC>LC weakest).

Ecosystem component		Fish		Benthos		Plankton		Seabirds	Marine mammals		Habitat		
		Fish (F)	Cephalopods (C)	Benthos (B)	Macrophytes (M)	Zooplankton (Z)	Phytoplankton (P)	Seabirds (S)	Marine mammals (MM)	Marine reptiles (MR)	Water column & bio-chemical habitat (WH)	Physical habitat (PH)	Worst-case
Pressure													
Chemical	Synthetic compound contamination	LA	LA	LA	LA	LA	LA	WC	WA	U	LC	N	S, MM, MR
	Heavy metal contamination	LA	LA	LA	LA	LA	LA	WC	WC	U	LC	N	S
	Hydrocarbon contamination	LA	LA	LA	LA	LA	LA	LA	WC	U	LA	N	S
	Radionuclide contamination	LC	N	LC	LC	U	U	WC	WC	U	N	N	F, S, MM
	Changes in nutrient levels	LC	LC	LC	LC	LC	LA	LC	N	N	LA	N	P, WH
	Changes in salinity	LC	LC	LA	LA	LC	LA	N	N	N	LA	N	B, P, M, WH
	Changes in oxygenation	LA	LA	LA	LA	LA	LA	N	N	N	LA	N	F, B, Z, P, M, C, WH
Biological	Introduction of microbial pathogens/ parasites	UC	UC	UC	UC	U	U	UC	WA	U	N	N	B, M, S, MM
	Introduction of non-native species & GMOs	WC	UC	WC	WC	WC	WC	WA	U	U	N	LC	B, M
	Selective extraction of species	WA	WA	WA	WC	LC	LC	WC	WC	WC	WC	WC	F, C, B

Widespread and chronic (WC) interactions are important when considering long-term changes at the scale of the North Sea and ‘Change in temperature’ as driven by climate change, is a concern for all biological components over these times scales. The contamination pressures (synthetic, non-synthetic and radionuclide) are almost all widespread in their interactions with the seabirds and marine mammals, whilst the ‘Introduction of non-native species and GMOs’ is thought to be a widespread concern for the fish, biological benthic and planktonic components. ‘Selective extraction of species’, although not thought to be acute in its effects, is still a widespread pressure on the macrophytes, seabirds, marine mammals, marine reptiles and both components of habitat (Table 4.2.5.1.).

Locally acute (LA) interactions, although less important at a regional ecosystem level such as that of the North Sea, will be important to take into consideration when carrying out an integrated assessment of a smaller subunit of the region. There are many more LA interactions (65) than there are WA interactions (10) (Table 4.2.5.2). Key pressures that result in such interactions include many of the physical and chemical pressures, for example, the contamination pressures and the pressures related to substratum loss, smothering and changes in thermal regime. They are particularly relevant to the ecosystem components that are sessile or low motility in nature such as the benthos and macrophytes. Locally chronic (LC) interactions may be important for long-term changes in sub-regional areas picked up by time series for local monitoring programmes.

Other major patterns to emerge from an analysis of the data are from a number of the ecosystem components, most notably the ‘Physical habitat’, where there are only a few pressures, whilst for others components there are interactions with many pressures. These often vary in their geographical extent and intensity (see summary in Table 4.2.5.2).

Table 4.2.5.2 Summary of the number of pressures applicable to each of the ecosystem components at a regional North Sea and local spatial scale.

	Widespread (North Sea)		**Local scale**	
Ecosystem component	Acute	Chronic	Acute	Chronic
Fish	1	2	6	10
Cephalopods	1	1	9	6
Benthos	2	2	8	7
Macrophytes	2	2	10	5
Zooplankton	0	2	5	9
Phytoplankton	0	2	9	5
Seabirds	1	5	3	3
Marine mammals	2	5	2	2
Marine reptiles	0	2	2	2
Water column and bio-chemical habitat	0	2	8	5
Physical habitat	1	1	3	2

4.2.6 Metrics, Indicators, Data series and Reference levels for Key Pressures

4.2.6.1 Introduction

Restricting ourselves to those pressures that have been found to be widespread and acute in their interactions with particular ecosystem components – the key pressures, we have considered whether there are available metrics that could be used as possible indicators of these interactions (summarised in Table 4.2.6.1 below). These indicators should describe the pressure specific to the individual component. In section 4.2.6.2 we give a brief description of the utility of these metrics as indicators of key pressures based on whether there are available data series and reference levels, and whether they have been found to meet the ICES criteria (ICES, 2001) for good indicators. In all cases, where we refer to reference levels we mean the OSPAR definition which is “the level of the *metric* where the anthropogenic influence on the ecological system is minimal”.

Table 4.2.6.1. Possible indicators for the key pressures of those ecosystem components where a widespread and acute (WA) interaction was reported in Table 4.2.5.1. An asterisk (*) indicates where we know that metrics resembling those described as possible indicators here, are currently being developed or considered in other fora.

Key Pressures (WA)		Ecosystem component	Possible Indicator
Physical	Abrasion/ physical disturbance	Benthos	Abundance of sensitive taxa*
		Macrophytes	Percentage cover of sensitive taxa
		Physical Habitat	Area of extent of highly sensitive or threatened habitats*
Chemical	Synthetic compound contamination	Marine mammals	Body burden/tissue concentrations*
Biological	Introduction of microbial pathogens/ parasites	Marine mammals	Pathogen/parasite loadings of populations*
	Introduction of non-native species and genetically modified organisms (GMOs)	Seabirds	Proportion of eggs destroyed by non-native species at a population level
	Selective extraction of species	Fish	Landings of commercial species*
			Fisheries mortality (F)*
			Modelled mortality of species and communities*
			Changes in the proportion of large fish and hence the average weight and average maximum length of the fish community*
		Cephalopods	Landings of commercial species*
			Modelled mortality of species and communities
		Benthos	Landings of commercial species*
			Fisheries mortality (F)*
			Modelled mortality of species and communities*

4.2.6.2 Descriptions of the possible indicators

- Abrasion/physical disturbance

Abrasion/physical disturbance will impact primarily on physical habitat, benthos and macrophytes through direct mortality, indirect effects on fitness, and alteration of the structural properties of habitat. Indicators that could be used relate to changes in the abundance of the living components and in particular taxa identified as being sensitive to the physical disturbance. For physical habitat features, indices based on the area of extent of sensitive or threatened habitat would serve.

Sensitive benthic taxa, and, by extension, similarly identified macrophyte species, have previously been suggested to be potentially good indicators of the physical disturbance due to trawling (section 3). It therefore seems likely that the same approach could be extended to other forms of abrasion/physical disturbance. However, the process for rigorous selection and testing of such sentinel taxa has shown that very few meet the criteria of good, tightly coupled, indicators (ICES, 2004b). WGECO has previously advised that given current levels of knowledge of the distribution of seafloor habitats and the absence of an agreed framework for the assessment of benthic habitat quality (ICES, 2005b), further development of indicators relating to these properties is likely to take some time, particularly at the scale of the North Sea.

It will not be possible to set OSPAR defined reference levels (e.g. no abrasion/physical disturbance) for these indicators because there are no time series for any of these components that date back to a period when abrasion/physical disturbance were not widespread (as a result of bottom trawling) in the North Sea. We do not yet have complete maps of physical habitat or macrophyte cover and North Sea scale distributions of benthos are only available for a number of discrete time periods (see section 8, this report). It may be more fruitful to explore new modelling techniques that use hydrographic and physical features such as bottom currents, shear stress, depth, storm frequency, to parameterise models of expected reference conditions for the biological and physical elements of habitat, which includes all three components described above.

- Synthetic compound contamination

Synthetic compound contamination is a key pressure for marine mammals where toxins can be bio accumulated to lethal levels leading to mortality. Although this has been described as a widespread and acute pressure for marine mammals, it is in reference to a limited number of examples over the last 10 years, where particular populations suffered acute effects.

Indicators that could be used relate to measures of the body burden or tissue content of synthetic compounds taken from impacted populations. These have never been formally tested by WGECO as indicators of pressure but they would match the ICES criteria in terms of being closely matched to the pressure. The working group on Marine mammal ecology (WGMME) could provide useful information in terms of the suitability of such indicators.

As it is now possible to use non-intrusive methods to take body tissue samples from marine mammals it should be possible to monitor levels particularly in those groups that spend part of their lives on the shore (e.g. seals). For the cetaceans it may be more difficult to monitor and strandings may be the only method possible. Reference levels as defined by OSPAR do exist and these refer to natural levels of synthetic compounds found in body tissue. Again WGMME may be able to advice on reference levels.

- Introduction of microbial pathogens/parasites

The introduction of microbial pathogens/parasites has been described as a key pressure for marine mammals. This refers specifically, however, to the worst-case scenario, whereby an introduced microbial pathogen, Phocine distemper virus (pdv), spread rapidly, attacking the immune system of individuals. It was responsible for the deaths of a significant proportion of the harbour seal (*Phoca vitulina*) population in the North Sea (Teppema et al., 1990; Swinton et al. 1998; Jensen et al. 2002).

Indicators that could be used to monitor such outbreaks include pathogen/parasite loadings at the population level for species of concern. These have never been formally tested by WGECO as indicators of pressure, but could match the ICES criteria in terms of being closely matched to the pressure, if the technology is available to measure the actual pathogens or parasites. Further work is required to assess whether the available technology will allow for the development of such indicators.

In terms of data series and reference levels, the same method applies as described above for synthetic compound contamination and again consultation with WGMME would be useful.

- Introduction of non-native species and genetically modified organisms (GMOs)

The introduction of non-native species and GMOs has been described as a key pressure for seabirds. However, this is based on the worst-case scenario and refers only to introduced non-native species, not GMOs. A number of seabird species have suffered acute effects at the population level due to the introduction of rats, which have seriously impacted breeding success due to their predation of the birds' eggs.

The indicator we suggest that could be used to monitor the interaction of this pressure with seabird populations is the proportion of eggs destroyed by non-native species at a population level. This indicator have never been formally tested by WGECO as an indicator of pressure but it may not match the ICES criteria in terms of being closely matched to the pressure, because it may not be possible to tell which species has destroyed the eggs. The working group on seabird ecology (WGSE) may provide useful insights into the utility of this as an indicator.

Regular surveys of nesting seabirds are undertaken and it should be possible to count how many clutches of eggs are destroyed by land-based predators. Reference levels would be related to those where only native predators are found and again consultation with WGSE would be useful.

- Introduced GMOs

It is unlikely that GMOs will have any widespread acute interactions with any of the ecosystem components; rather, interactions are chronic, acting through effects on wild populations through inter-breeding, competition between GMOs and wild populations and disruption to reproductive cycles (ICES 1995, 1996, 1997, 1998). In terms of identifying indicators to monitor the interaction between introduced GMOs and wild stocks, some difficulty may be experienced and the ICES WG on the Application of Genetics to Fisheries and Mariculture (WGAGFM) should be consulted.

- Selective extraction of species

Selective extraction of species is a key pressure for the fish, cephalopods and benthos (in particular commercial species) through direct mortality to target and non-target species. In addition this mortality is size-selective at the population and community level when related to fisheries, which is the main activity that contributes to this pressure in the North Sea. At this stage we do not suggest indicators for this pressure that cover total 'Selective extraction of species'. These would need to include measures of removals from recreational fisheries, aquaculture and research-based removals.

In all cases, landings of commercial species can be used as an indicator of this pressure. However, we note caution in terms of the potential non-linear relationships between reported landings and actual total fisheries mortality (see section 8 this report). Also, this will not account for mortality to the non-target species or to the discarded element of the catch. Fisheries mortality (F), as used in the stock assessments, does take into account some unaccounted mortality for discards but this is not the case for all commercial stocks and data are only available for the fish. At this meeting, WGECO have presented the work of some new modelling analyses that quantify fishing mortality to the fish and benthos components at the community level, based on the distribution of fishing effort and the vulnerability of the different taxa that make up impacted communities (section 3.10 and 3.11). These modelling approaches could be explored further in terms of potential indicators of total extractions of species from the North Sea.

Landings data and fisheries mortality values are available for commercial stocks and reference levels relative to OSPAR are ultimately zero, although in reality, functional targets would need to be set in relation to social and economic objectives. In terms of collecting the data to model fisheries mortality on the benthos, fish and cephalopods, effort data and abundance data are required at the North Sea level. Regular surveys are undertaken of the fish communities, although we recognise that the survey methods do not provide a complete record of the absolute abundances and distributions of all species. Similar surveys are not undertaken for the benthos or cephalopods at this time. Reference levels would need to be set based on the same issues discussed for landings and fisheries mortality.

4.3 Defining the uses of the IA framework

4.3.1 Introduction

There are two different applications for which this approach could be used. The first relates to the requirement to interpret signals detected by Integrated Assessments (IAs) or ‘State of the Seas’ reporting. Such reporting is intended to highlight temporal trends and/or spatial patterns for which further explanation is needed; specifically;

- are there any consistent trends in ecosystem status?
- if so, which human activities are likely to be contributing to these trends?

By matching which indicators are changing to the pattern of weights of various pressure-state combinations, it should be possible to provide advice to Government Departments or Client Commissions on several points informative to management, such as:

- which key pressures of human activities are likely to be responsible for the observed trends or patterns in the indicators (and the ecosystem which they are being used to assess.)?
- which human activities are likely to be producing the specific mix of pressures?
- what would be an appropriate set of indicators by which these pressures, and the activities responsible for them, could be monitored most efficiently in future (section 4.3.2)?

The alternative application would be relevant to specific sectoral managers and the corresponding industries which require an understanding of their impact on the marine ecosystem and advice on indicators that might be suitable for assessing their effects. Many of the steps would be similar to the first usage, but done with different emphases. The intention would be to identify a (hopefully) small set of indicators by which the specific mix of pressures characteristic of that sector’s impacts could be monitored.

4.3.2 Using indicators within these approaches

Indicators are widely used for environmental reporting and management support and they will have an important role in the application of any assessment based on this IA framework (ICES, 2005b). The identification of appropriate indicators depends on the purpose for which they will be used. To provide input to broad scale environmental reporting, components and their attributes are selected to be representative of the ecosystem, and it may often be necessary to select several indicators to track the state of one component and attribute, or one indicator may track the state of several components and attributes (Jennings, 2005). For comprehensive environmental reporting, descriptive state indicators would be selected to cover all components listed in Table 4.2.5.1 taking account of their relevant attributes as illustrated by the matrix shown in Figure 4.3.2.1.

Where specific human activities result in adverse impacts to the ecosystem and management wished to reduce or reverse these impacts, it will be necessary to use indicators (performance indicators according to previous WGECO definitions; ICES 2004) that are specific to the primary pressures of those activities and which have the characteristics of good indicators as outlined in previous WGECO reports (ICES, 2001, 2004 and 2005b). In particular, we recognise that managers can only manage activities. The mechanism that links state of the ecosystem components or attributes to the managed activity must be understood if performance indicators are to be utilised. Ideally, one or several such indicators would be used to monitor the mitigation of the impact and achievement of the objective, measured by comparing the values or trends in the indicator with a target value, reference direction or trajectory (FAO, 2003).

Attributes

Components

	Abundance	Diversity	Trophic structure
Population (1,2 ….. n)			
Species (1,2 ….. n)			
Community (1,2 ….. n)			

Figure 4.3.2.1. Matrix of the breakdown of components and example attributes that could be used to identify the indicators required for environmental reporting or management (Taken from Jennings, 2005).

4.4 Using indicators of key pressures to investigate change in state

4.4.1 Approach

The approach taken in Table 4.2.5.1 to identify the extent and intensity of the links between pressures and ecosystem components is an important step in the process of identifying how manageable activities may have caused observed changes in ecosystem state in an integrated assessment. The strength of the interactions focuses attention on defining a suite of indicators for particular pressures that have resulted in a change in state of one or more ecosystem components (4.2.6). In order to investigate how key pressures have contributed to a change in state the following steps would be followed:

i) Undertake an overview assessment of the status and trends in key state variables describing the ecosystem components. This will describe attributes at the species, population and community level.

ii) Having noted any adverse changes in these variables, consult Table 4.2.5.1 to identify the key pressures that may have contributed to these changes. This may require the examination of multiple state indicators and an understanding of the interactions between them. If available, specific pressure indicators that describe the interaction between a particular pressure and the ecosystem component may contribute to this interpretation (Table 4.2.6.1).

iii) Identify which activities could have contributed to the key pressures identified in step (ii) by consulting Table 4.4.3.1. These pressure indicators should be examined for each of the contributing activities over relevant spatial and temporal scales.

Our approach assumes that there are available data and indicators to assess overall status and trends of all the ecosystem components in step (i). For the North Sea, this is not yet true for all ecosystem components (see section 4.4.4 below), so when illustrating the application of this approach we restrict ourselves here to the fish component, for which there are available indicators of state and routine data collected to support them.

4.4.2 Linking change in state to key pressures (steps (i) and (ii))

In 2005, WGECO compiled a list of potential state indicators (ICES, 2005b, section 6). This list has been shortened to reflect only those applicable to the fish component and is given below (Table 4.4.2.1). General trends in state of the fish component could be assessed using a selection of these indicators to represent the key attributes of species, population and community level state, but appropriate criteria will be required to identify the most suitable subset. Long-term time series are available for trend analysis of the fish component in the North Sea (see section 9, ICES, 2004). It is well known that changes have occurred in both the fish community and the populations of particular species of fish (Heessen & Daan 1996; Piet & Jennings 2005; Greenstreet & Rogers 2000, 2006). Some of these changes are considered to have been driven by anthropogenic activities and Table 4.2.5.1 will identify the key pressures on the fish component at the North Sea level.

At the scale of the North Sea, widespread pressures should be considered first. For the fish component, this is restricted to ‘Selective extraction of species’ (scored as ‘widespread’ ‘acute’). Fisheries Mortality is already available for the fish component to monitor ‘Selective extraction of species’ at the population level for commercial species and we have suggested a number of other potential indicators in section 4.2.6. We also acknowledge that ‘Change in temperature’ due to climate change and ‘Introduction of non-native species and GMOs’, although chronic in nature, are widespread, and indicators that reflect the impact of change in these pressures on the fish component will need to be considered ultimately.

At the local scale, many more pressures affect the fish component (Table 4.2.5.2) and further work will be required in terms of developing tools for considering the effects of multiple pressures on ecosystem components.

Table 4.4.2.1 Potential indicators for Ecosystem State of the Fish component (modified from ICES, 2005b).

Ecosystem Element	Subset	Aspect	Indicator
Population	Assessed	Status stock	• Recruitment (R) • Spawning Stock Biomass (SSB) • Total mortality (Z) • Mean Age of the population
		Health	• Condition factor • Incidence of disease, pathogens, parasites, contaminants • Genetic diversity
	Non-assessed	Status species	• Total Biomass • Total Number • Presence of indicator, charismatic, sensitive species
		Health	• Condition factor • Incidence of disease, pathogens, parasites, contaminants • Genetic diversity
Community	Size structure	Abundance	• Slope size-spectra • Mean weight or Mean length • Proportion of large fish • Length-frequency distribution • *k*-dominance curves • Multi-dimensional ordination
	Species composition	Abundance	• Species presence / abundance • Index of rare species • Index of declining or increasing species • Proportion of sensitive or threatened species • Presence of Non-indigenous species • Species turnover/loss rates • Theoretical Distribution Metrics Log-Series and Log-Normal • *k*-dominance curves • Multi-dimensional ordination
	Species composition	Life-history	• Mean maximum length • Size above which 50 % of the population is mature • Mean maximum age • Age above which 50 % of the population is mature • Fecundity expressed as number of eggs per female or number of eggs per body weight • Mean k and/or L_∞ of von Berthalanffy growth curve
Community		Biodiversity	• Hill's N0 N1 N2 • Species-Effort Index • Taxonomic Diversity Indices

Table 4.4.3.1. The relationship between activities in the North Sea and the mechanisms through which they exert pressure on the marine ecosystem. Shading indicates an existing mechanism for a particular activity, while '?' indicates a potential mechanism.

	Activity	Aquaculture	Climate change	Dredging	Oil & Gas	Renewable energy	Aggregate extraction	Fisheries	Recreation	Military	Research	Shipping	Land-based discharges & emissions
Pressure													
Physical	Substratum loss			■	■	■	■	■			■		
	Smothering	■		■	■	■	■	■			■		■
	Change in suspended sediment	■		■	■	■	■	■	■		■		■
	Change in water flow rate	■	■		■	■	■	■			■		■
	Change in temperature		■		?							?Ballast	■
	Change in turbidity	■	■	■	■	■	■	■	■		■		■
	Change in sound field		?	■	■	■	■	■	■	■	■	■	
	Change in light regime	■	■	■	■	■	■	■			■		■
	Visual presence	■			■	■		■	■		■		
	Abrasion/ physical disturbance	■		■	■	■	■	■			■		
Chemical	Synthetic compound contamination	■		■	■		?			?		■	■
	Heavy metal contamination			■	■		■	■		■	■	■	■
	Hydrocarbon contamination			■	■		■	■	■	■	■	■	■
	Radionuclide contamination			■	■		■	■		■	■		■
	Changes in nutrient levels	■	■	■	■		■	■	■		■		■
	Changes in salinity		■									■	■
	Changes in oxygenation	■	■	■	■	■	■	■	■		■		■
Biological	Introduction of microbial pathogens/ parasites	■	■					■	■	■	■	■	■
	Introduction of non-native species & GMOs	■	■		■			■	■	■	■	■	
	Selective extraction of species	■						■	■		■		

4.4.3 Identifying activities contributing to key pressures (step (iii))

In order to identify which of the manageable human activities may have contributed to the pressures identified, it is necessary to refer to a second table which links the recorded human activities for the ecosystem of interest with pressures (Table 4.4.3.1). Given that 'Selective extraction of species' is the key pressure for the Fish component, it is possible to identify that the main activities contributing to such a pressure are Aquaculture, Fisheries, Recreation and Research. Indicators that describe the spatial and temporal variability in intensity of these activities in relation to the pressure 'Selective extraction of species', should be consulted. A number of potential indicators are listed for Fisheries in Table 4.4.3.2.

Table 4.4.3.2. Pressure indicators that relate to commercial fishing activity and the pressure 'Selective extraction of species'.

Activity	Indicator
Fisheries	• Total catch • Total landings • Total discards • Total fisheries-induced mortality or direct mortality • By-catches of protected species and discards

4.4.4 Comment on preparedness to undertake comprehensive assessments of ecosystem state in the North Sea

In ToR (f) (section 8 of this report), WGECO have reviewed work undertaken by the Regional Ecosystem Group for the North Sea (REGNS) to compile and assess data available through the ICES working group structure to undertake an integrated assessment of the North Sea. It is notable that a number of the ecosystem components included in the IA framework here, are not represented at all in the REGNS assessment. These include macrophytes, marine mammals, marine reptiles and the physical habitat. Others are only represented by very low numbers of data points that would preclude any useful assessment of status and trends at the North Sea scale (e.g. benthos and cephalopods), whilst others are only available for limited spatial extent (e.g. water column and bio-chemical habitat) (see section 8, Table 8.2.1.1).

In 2004, WGECO considered how data routinely collected in the North Sea could be most effectively utilised for the purpose of reporting on ecosystem state (ICES, 2004b section 9.3). The assessment suggested that all ecosystem components could be covered by routine data collections (excluding macrophytes, marine reptiles and cephalopods which were not included at the time). However, the components acting as the primary objectives of these surveys were limited to fish, marine mammals and seabirds, with a limited number targeted at benthos, plankton and nutrients/habitat. Although the assessment by WGECO described secondary and additional data that could be and sometimes were collected in these routine surveys, the lack of comprehensive coverage by the REGNS approach suggests these data are not yet readily available, or in some cases, even collected.

It is important to build on the experiences gained by undertaking the REGNS assessment in terms of our preparedness to undertake comprehensive assessments of ecosystem state in the North Sea. In particular, we reiterate the potential (as outlined in ICES, 2004b section 9) for additional data collection that could be undertaken using existing routine surveys to increase coverage of those ecosystem components currently not well reported on.

4.5 Conclusion and way forward

We have completed the identification and selection of key pressures of human activities on the state of the marine ecosystem begun in 2005, based on their extent and intensity of interaction with each of the ecosystem components and grouped according to whether the intensity of pressure was widespread or local. This takes us further in our ability to select the interactions between components of the ecosystem and key pressures, and use this framework for indicator selection. However, in many cases, the performance indicators of components necessary for management do not or may never exist for some combinations of pressures and components. In this case, pressure indicators that track the actual activities must be monitored in association with environmental state indicators.

While WGECO are able to propose some indicators that could be used to summarise the effects of activities on the ecosystem, we recognise that the development of operational management indicators for many of the recommended categories will be a longer process and will need to be supported by significant further research. To summarise the effects of particular activities on the ecosystem, the indicators need to describe trends in components and attributes that are strongly influenced by the pressures attributable to those activities, even if the full mechanisms by which a particular pressure changes state cannot be described. WGECO recognise that the number of specific management indicators for the various activities that occur in the North Sea are very limited, but we are in a position to recommend state indicators that would support an assessment of the effects of the various pressures on the North Sea ecosystem.

There are important tasks still to be completed to advise on how an integrated framework might look, which together incorporates the requirements of the EMS to achieve good environmental status (GES), national and international objectives frameworks, and indicator initiatives of ICES, EEA and sectoral management (MSP) in Europe. The international drivers for this work are already well understood and govern the approach taken throughout Europe. Those within OSPAR Annex V (Biodiversity), the European Commission (EMS), and various nature conservation goals (Johanesburg, CBD, Bergen) will be influential, but must operate together and in a coordinated way to deliver sustainability in European seas.

Further work is necessary to bring these components, together with progress on indicators described above, together in an integrated framework for the provision of ecosystem advice within European Seas. Goals and objectives (described in (ICES, 2005b)) will influence the way in which specific ecosystem management activities are developed. Although these are clear in outline and well understood as a concept, there is still more work to be done to develop an objectives framework that corresponds with the ecosystem components and links to indicators described in this section. In particular, the current EcoQOs developed by OSPAR, and high level ‘headline’ indicators of biodiversity developed by other international fora, should be reviewed to identify their location in Table 4.2.5.1, and their suitability to form part of this larger framework.

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5 WGECO response to SGMAS progress

Examine and take forward recommendations of the Study Group on Management Strategies (SGMAS) meeting in early 2006) in their review of WGECO suggestions for ways in which ecosystem considerations could be incorporated into fisheries management strategies.

5.1 History and current approach

In its 2005 Report, WGECO (ICES 2005a) presented a detailed analysis of how ecosystem effects of fishing could be inserted into the approach to provision of advice using formal Management Strategies and Harvest Control Rules that had been proposed by SGMAS in its 2005 Report (ICES 2005b). SGMAS, in turn consider the WGECO proposals at its meeting, and commented on the WGECO proposals in section 10 of their report (ICES 2006a). This section of the WGECO Report takes this dialogue further in three steps.

- First, in Section 5.2 WGECO summaries its views on which ecosystem considerations should be taken into account in placing fisheries within an ecosystem approach, and how well the Management Strategies framework proposed by SGMAS can address each consideration. We highlight opportunities for immediate action to include the ecosystem considerations in the framework, and underscore the potential benefits of such action. Where the current readiness is low, we highlight the major weaknesses and consequent risks. Where we believe modification of the SGMAS framework would improve ICES ability to advise within an ecosystem context, proposals for the specific additions or modifications to the SGMAS framework are included in Section 5.3. Where knowledge or data are fundamentally inadequate at present, we clarify the type of research (including data collection and analyses and development of analytical methods) that needs to be undertaken to enable improvements in practice to be made.

- In what they consider their final Report, SGMAS (ICES 2006a) lays out its recommended ways forward for ICES to progress with building the ability to provide advice within structured management strategies. Because they consider this to be their last report, SGMAS tries to lay out nearly a step by step guide to how Expert Groups should proceed with developing management strategies for fisheries. If their Report is to be *de facto* the ICES Users Guide to Management Strategies, it is important that the necessary ecosystem linkages be pointed out as part of that guide. Hence, Section 5.3 works through the body of the SGMAS report, extracting many specific places where either environmental forcing on stock and fisheries dynamics, or impacts of the fishing on non-targeted ecosystem components, could be considered directly. The intent is both to highlight where opportunities to have Management Strategies either advised or used by ICES actually take seriously the commitments of ICES, member States, and client fisheries management agencies to apply an ecosystem approach.

- The direct response of SGMAS to WGECO's recommendations, in Section 10 of their report, includes questions as well as comments on our past points. WGECO replies to the questions posed in Section 5.4, and carries the dialogue between the two groups (and with the wider ICES community) further.

5.2 The Potential Ways that Ecosystem Considerations Should Be Included in Development and Application of Management Strategies

This section lists the major ecosystem-fishery considerations that have been of interest to WGECO, and highlights briefly where each one connects to the process SGMAS lays out for development, evaluation, and use of management strategies. More information on the nature of the linkages and how each area can be developed further can be found in the corresponding parts of Section 5.3.

Although this section focuses on the process of developing and testing management strategies, it is stressed that the starting process of setting objectives for management of any fisheries should include ecosystem considerations as well. This could include any aspects of the fisheries-ecosystem interactions, and as noted by SGMAS, should be undertaken before there is substantial investment in the development and testing of management strategies. WGECO's views on Ecosystem Objectives has been discussed thoroughly elsewhere (ICES 2002a), and will not be repeated here.

5.2.1 Effects of Environmental Forcing on the Exploited Resources

In the context of traditional assessment parameters that are also featured in Management Strategy computations, these effects are seen most readily in three places.

1) They can be seen as the influences of environmental conditions such as temperature, currents, salinity, and timing of events, on recruitment, growth and maturation of the targeted stocks. In the Management Strategy framework, these should be addressed in the first instance in the Operating Model, in the functional relationships used to model these biological parameters of the stock. Methods for doing so have been detailed in the work of SGPRISM (ICES 2002b) and WGGROMAT (ICES 2003), and are being carried further by the work of, for example, ICES-GLOBEC (as summarized in ICES 2004). However, these effects also represent an important part of uncertainty about future states of nature. Hence, where environmental forcing of population dynamics are possibly important, they must also be addressed explicitly in at least the robustness testing of alternative harvest control rules. Where such forcing effects may be systematic enough to require adaptable of harvest strategies to remain sustainable under different environmental conditions (e.g. environmental regime shifts [Steele 1998]), the Harvest Control Rules themselves may have to include switches or other complexities to ensure such adaptations are made in a timely manner.
2) They can be seen as the influence of those environmental factors on the distributions of targeted and bycaught species in the area being fished. Some of these influences can be captured in the biological processes parts of the Operating Model, in the same way as for effects of environmental factors on recruitment, growth, and maturation. Again, there may be challenges in specifying appropriate functional forms, and scientific progress on this issue is less mature than in the preceding point. Where species distributions do change substantially because of changes in environmental conditions, this is also likely to affect the mix of species taken as catches in different fleets. This has implications for the parts of the Operating Model dealing with fleet dynamics. Depending on the adaptability of the fleet, this could affect how their economic performance is modelled (e.g. through changes in fishing costs to find suitable catches), or what mix of fleets is optimal for taking the available yield from the target species (e.g. through changing the relative species composition of catches). Both of these factors would need to be in the Operating Model, and neither will be easy to represent. Moreover, both potential effects of changes in species' distributions in response to environmental forcers also become uncertainty about future states of nature. If they are important enough to be considered for inclusion in the

Operating Model, they must be considered explicating the robustness testing of various Harvest Control Rules as well.

3) They can be seen as mortalities due to predator- prey interactions that change as the size (and possibly species) composition in the sea changes. Some of these interactions have been captured in analytical assessments for many years, either dynamically (e.g. MSVPA) or as predator "biomass reserves" and are readily incorporated into Operating Models. Where such interactions are considered important and highly variable over time, they may be appropriate to include explicitly in harvest control rules as well, either dynamically or as switches. In such cases it is also important that they are also considered in robustness testing of management strategies, whether they end up as part of a complex control rule or not.

5.2.2 Effects of the Fishery on the Ecosystem

There are four major aspects of the impacts of fishing on ecosystem features that are directly relevant to the development and testing of Management Strategies. Each is a class of issues that is diverse and complex, and in aggregate bring biodiversity considerations squarely into the ICES advice. ICES is differentially prepared to make the necessary linkages between the four types of ecosystem issues and the management strategy process, and this affects the nature of the proposal that are made for immediate actions as developed in Section 5.3 and for research priorities and initiatives within ICES.

Bycatches: The direct mortality cause by fishing on non-target species can be a major ecosystem consideration. This is particularly the case when a bycatch species can sustain only a lower exploitation rate than the target species, is the case for marine mammals and may be the case for some elasmobranches, for example (Walker and Hislop 1998, Dulvy et al. 2000). This ecosystem effect of fishing can be incorporated directly into the current approach to management strategies for mixed-species fisheries. WGECO illustrated how at least starting estimates for sustainable mortality rates of bycatch species can be estimated in ICES (2001). Combined with survey-based estimates of population sizes of such species and reliable information (observer-based if possible) on total catch composition of each fleet, the bycatch species just become additional species in the mixed-species fisheries Harvest Control Rules. Where data on the actual species composition of the catches are available then the maximum sustainable "harvests" can be estimated well enough to be added to the list of catch constraints on fleet-by-fleet effort. Then existing algorithms for solving for the allocation effort among fleets that gives maximum catches without overexploiting any species, can give solutions which take bycatches into account. Which species to include in these more complex Harvest Control Rules is driven by the full suite of Ecosystem Objectives for the area. However, presently the necessary data on full catch composition are not available for many fleets, so this very practical approach to an important problem will be able to be implemented readily.

Trophic Relationships: Any selective fishery necessarily affects trophic relationships through changing the relative abundances of predators, prey, and competitors, and when fisheries are removing substantial portions of biomass of at least some size groups, these effects can be large and pervasive. Hence, how these are addressed in the management strategies process depends greatly on the suite of Ecosystem Objectives set for the system. If the objectives are vague and general, accommodating all possible trophic effects in fisheries management strategies can become intractable or excessively time-consuming to address. Where specific trophic relationships are known or suspected to be strong, they can be incorporated explicitly in the Operating Model, and many methods such as MSVPA exist for doing so. Where trophic relationships have much more of a network structure, so individual relationships may be inappropriate to include explicitly but their aggregate effect is still important, more complex Operating Models might include the fishery impacts on ecosystem properties such as its size spectrum or maximum size of the fish in the community. Such a strategy would probably work best for Operating Models of major multi-species fisheries, but at this point such ideas are speculative. There has been too little exploration of this aspect of ecosystem impacts of fisheries to be confident of the best way to represent it in the Operating Model. What can be said is that the more important these predator-prey dynamics appear to be in the Operating model, the more important it is to include them in the robustness testing as well. Again, however, the scientific community has made few proposals for how to conduct robustness testing with regard to this source of uncertainty.

Where there are explicit Ecosystem Objectives related to trophic relationships, it will also be necessary to include some indicator of the status of the key species (or perhaps at least its relative status compared to other key linked species) in the Harvest Control Rules being tested. That enables both the harvest to be adjusted as the status of the trophically key species changes, and the performance of the Harvest Control Rule relative to the corresponding Ecosystem Objective to be evaluated. There seem to be several cases histories on which this approach can be built (Section 5.2.4), when the trophic relationship of concern is a bottom-up one, such as reserving adequate biomass for dependent predators on a fished stock. Where the important trophic relationships are thought to be top-down, it is less clear how to accommodate these in Harvest Control Rules. Trophodynamic models might make some predictions about how different harvest strategies on the controlling predators affected the likelihood of achieving Objectives for the prey species or food web as a whole. However, WGECO has reviewed trophodynamic models several times, and found few that were considered sufficiently robust for use in management applications (Rice 2005). There are major data limitations in parameterising trophodynamic ecosystem models. Moreover usually multiple trophodynamic models with functional forms different enough to perform very differently in Operating Models can be fit equally well to what data are available. More work is clearly needed in this area. WGECO considers that there may be important lessons to be learned from the growing importance of "ensemble forecasts" in climate change research (http://www.metoffice.gov.uk/corporate/scitech0304/ensemble.html; http://www.ipcc.ch/ , where policy and management required forecasts, of the consequences of alternative management actions and had to consider similarly complex and uncertain relationships.

Habitat Impacts of Fisheries: Like the other ecosystem effects of fishing, impacts of fishing activities on marine habitats will enter the management strategy process first through the Ecosystem Objectives that are set for habitat protection. However unlike objectives for bycatch species and trophic relationships, that at least often can be expressed as population (or multi-population) objectives that fisheries are used to addressing, these objectives will often be set as spatially-based objectives rather than population-based ones. Then putting spatial components into the Operating Model, harvest control rules, and robustness testing all become new challenges for ICES. Some of the pieces to begin meeting this challenge do exist. The STECF data-base on catches has been developed at the scale of ICES rectangle. This database allows some spatial dynamics to be placed in the Operating Model, for both the stocks being harvested and the behaviour of the fleet. Some models have been developed that explore the consequences of redistribution of fishing effort among rectangles (e.g. Hutton et al. 2004) and from these alternative Harvest Control Rules for spatial allocation of effect by fleet can be built. This type of work on spatial management approaches has not developed very far in management strategy evaluation frameworks, but may get a major push from the impending EU Marine Strategy Directive, which is expected to give major emphasis to space-based issues (http://europa.eu.int/comm/environment/water/marine.htm).

However, it is becoming clear that many spatial questions that will have to be addressed in Objectives and associated Management Strategies will occur at spatial scales below the scale of ICES rectangles. Examples include the 2005 advice on spatial management of the Rockall Bank fishery to avoid *Lophelia* beds (ICES 2005c), the WKFMMPA (ICES 2006b) workshops and associated project intended to provide of science support for managing fisheries in some Special Areas for Conservation (SACs) identified under the EU Habitats Directive and Special Protection Areas (SPAs) identified under the Birds Directive. These spatial management questions are being posed at scales of hundreds of meters to a few kilometres, rather than on scales of 30 nautical miles or greater. There are major challenges to ICES in developing the science capacity and tools to explore the effects of alternative harvest control rules at such spatial scales. Moreover, as the spatial scale of the management questions being addressed in the management strategies changes, so will the perceptions of the ecosystem issues that should be addressed.

Regarding the spatial scale of the work, there is also a need to develop approaches and analytical tools for robustness testing to spatial management measures. Both biological uncertainties and fleet uncertainties exist in space as well as at population scales. These become very challenging when the fisheries management system includes consideration of multiple fleets, and the fleets operate in different areas or express their dynamic responses to stocks and regulations on different spatial scales themselves. Much new work will have to be undertaken by ICES and the larger scientific communities, to provide the scientific support necessary to manage human activities in the sea on the basis of spatially-based measures as core management tools rather than the traditional population-based measures.

Genetic makeup of populations: WGECO has highlighted that population-based management can allow fisheries to selectively deplete the genetic diversity of a population (Rice 2005). Again, Ecosystem Objectives with regard to genetic diversity of the target species can bring these considerations into the Management Strategy process. Then many of the considerations just discussed for taking account of habitat impacts of fishing within the development and evaluation of harvest control rules and management strategies would apply here as well. However, in the case of reducing detrimental effects of fishing on habitat features, it is expected that the search would be for harvest control rules that would remove fishing from areas that are identified as ecologically or biologically significant. In the case of protecting genetic diversity of target species, the search would be for harvest control rules that would spread fishing effort widely and often proportionately to the distribution of the target species. These two types of objectives may therefore sometimes be in conflict, presenting yet another set of trade-offs and choices to be addressed in scientifically based management strategies.

5.2.3 Conclusions and Recommendations

1. The methods developed or reviewed by SGPRISM and WGGROMAT should be included in Operating Models as a *routine part of any management strategy evaluation*. This should be done in ways that ensure the evaluation addresses both the effects of major environmental forcers in modelling of the population/system dynamics and in testing robustness of any harvest control rule against environmental aspects of the uncertainty about future states of nature.

2. The suite of objectives to be achieved by *any* management strategy being developed and evaluated must include ecosystem objectives appropriate to the major likely impacts of the fishery on non-target components of marine ecosystems.

3. Comprehensive, reliable, and scientifically accessible data *must* be made available on the full species and size composition of catches of all fishing fleets used in “harvest allocation algorithms” that form part of management strategies or harvest control rules for multispecies fisheries. Harvest constraints for non-target species considered at risk of non-negligible impacts from bycatches should be estimated using the best scientific information and methods available, and included in the vectors of catch constraints when these algorithms are applied.

4. ICES and member states should expand their scientific efforts to develop tools to include trophodynamic relationships and consequences in management strategy evaluations. These are needed both in terms of modelling system and population dynamics within the Operating Model, and in including trophodynamic considerations in robustness testing relative to future states of nature.

5. ICES and member states should undertake studies to greatly expand the scientific basis for including spatial management tools as components of management strategies. These efforts should consider spatial aspects of fleet-target-species relationships, fleet-non-target-species relationships, and fleet-habitat interactions.

5.3 Specific Points in the SGMAS Report

The important ecosystem considerations that need to be incorporated in the development and evaluation of management strategies are covered in Section 5.2 of this report. However, that Section is around major ecosystem issues, and maybe hard to link to specific steps SGMAS proposed for the process of developing management strategies. Consequently, the following Section walks sequentially through the process outlined by SGMAS, and comments wherever appropriate on how the various ecosystem considerations can be linked to their processes.

5.3.1 Conceptual Issues (their Section 2)

We note the central importance of Fig 2.1 in their document (presented below as Fig. 5.3.1.1) as their central conceptual approach to Management Strategies. Although there are important ecosystem considerations in the components of the flowchart addressing the "adaptation system" of the actual fishery, we acknowledge that the two components of the "Fishery System" with primary relevance to the concerns of WGECO are the "knowledge production system" and the "management decision system". The goal is to ensure that ecosystem aspects of fisheries are acknowledged as relevant considerations in the development of all Management Strategies, and only dropped from explicit representation in models and analyses after determining that dropping them does not increase risk, rather than only adding them explicitly when some ecosystem factor has been shown to be of exceptional prominence. That is, we feel addressing ecosystem considerations should be "business as usual" in development of Management Strategies, rather than afterthoughts or *post hoc* fixes to problems which crop up. The narrative following the figure in the SGMAS report does not suggest that ICES is there yet.

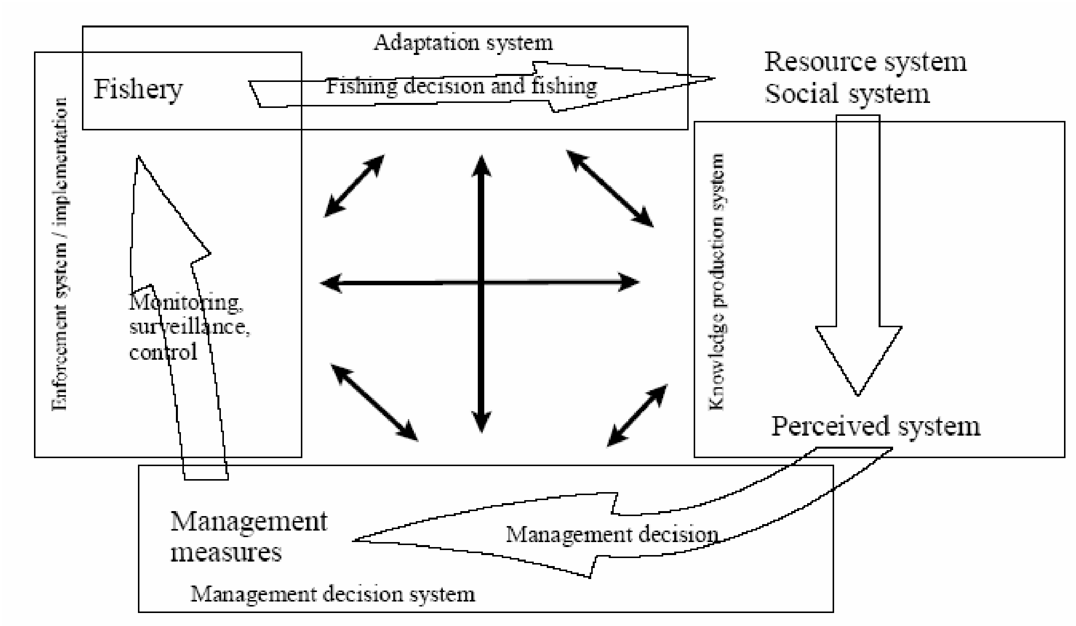

Figure 5.3.1.1. The fisheries system. The management strategy identifies the knowledge production system, the management decision system and the implementation system. The adaptation of the fleets and the natural changes in the resource system are external constraints. (From ICES 2006a).

Correspondingly, we note that where they correctly assert “The evaluation of management strategies is not a simple task. In general, the evaluations of management strategies are likely to involve analyses that go beyond the natural sciences which traditionally have defined ICES’s role. ICES should either attract this wider disciplinary perspective or should seek cooperation with their organisations”. We stress that even within the “traditional disciplines” of ICES, there is significant scope for attracting wider ecosystem perspectives to the fisheries advisory tasks.

5.3.2 Options for Management Objectives

It is conspicuous that in Section 3.1.2 of their report, SGMAS discusses the “Types of Management Objectives”, and covers a wide range of potential objectives for fisheries; biological, social, and economic. However in their discussion of biological objectives, every single one of them is about status and uses of target species in the fisheries. When ICES communicates with partners about the development of management strategies, it is essential that at least some ecosystem objectives be made part of that dialogue from the beginning.

The other important point for WGECO in Section 3 of the SGMAS report is their Figure 3.1, the “classic” three-stage harvest control rule” for biomass and fishing mortality [presented below as Figure 5.3.2.1]. WGECO notes that at least as a starting point, this “classic” rule could work for ecosystem attributes that have met our criteria as valid operational ecosystem indicators for fisheries management. Section 4 of this report discusses how to determine which of the ecosystem indicators that meet our overall criteria for scientific soundness (Rice 2005) are also valid for use in management contexts. Much of the SGMAS Report is extending this classic framework to be more robust and reliable, and at least the portion of the extensions dealing with improved representation of biological and bio-physical processes and relationships will be as applicable to other ecosystem properties as it is to the target species.

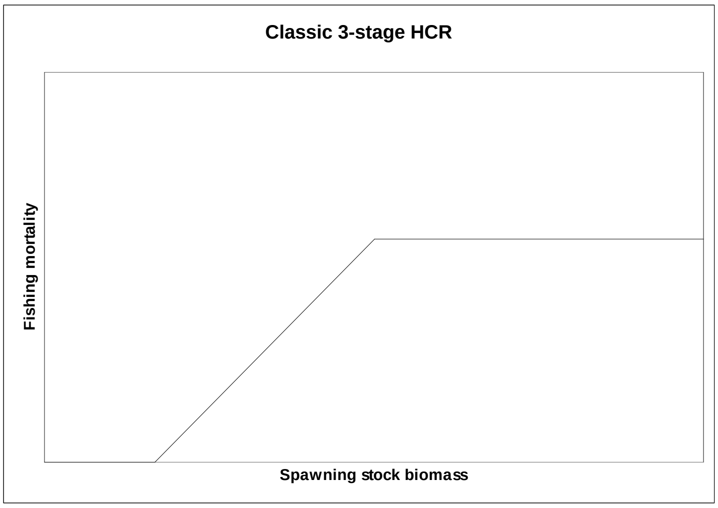

Figure 5.3.2.1. A "classic" three-stage HCR with specified, usually fixed, values for F when B is below the lower trigger point or above the upper one, with a smooth transition at biomass values between the two trigger points (From ICES 2006a).

Two qualifications have to be added when this "classic" model is applied to ecosystem indicators rather than target species SSB however.

1. The determination of biologically sound reference points, which are the inflection points on the figure, has proven challenging for SSB and F of target species. Several ICES Expert Groups, including WGECO, have been working to produce reference points for other ecosystem systems as part of the OSPAR EcoQO framework. Challenges in identifying references points for ecosystem indicators are usually greater than for SSB and F of target species, because both data and biological studies of underlying functional relationships are usually less complete. However, the challenge is proving tractable for many indicators, so there is scope for rapid progress, at least in some areas.

2. The SGMAS Report states that "Evaluation of these types of HCRs is relatively well understood and involves simulation". Whereas WGECO agrees that for ecosystem indicators the evaluations will involve simulation, it is less confident that there is a comparable understanding of how to evaluate many aspects of HCRs when functioning in contexts of ecosystem approaches to management. The technical evaluation approaches *may* be the same as for HCRs for target species, but the issue has not yet been explored fully enough to be confident that evaluation of HCRs is "relatively well understood" for all ecosystem considerations. The progress made on this aspect of the overall task by the MSE project in Australia illustrates that these problems are tractable with current amounts of data and scientific understanding of marine systems (Fulton, et al., (2005); Little et al., (2006)), even though performance would undoubtedly be improved by more and better data, and better understanding of relationships for most or all parts of marine ecosystems and human uses of them.

5.3.3 Evaluation of strategies

On a positive note, WGECO is pleased to see that in Section 4.4.2, SGMAS explicitly acknowledges Ecosystem Objectives as a key general consideration in evaluation of management strategies, on par with socio-economic objectives, the precautionary approach, and sustainability. However this inclusive view is not carried through the rest of the section. In particular, in introducing this topic, SGMAS says "the development and evaluation of harvest control rules needs to take place through an ongoing dialogue between ICES and the client fisheries managers", and their first guideline for this interaction is "Candidate HCRs should be identified by fishery managers and ICES in a dialogue process." WGECO repeats that environmental and nature conservation managers have a legitimate place in this dialogue process, and the makeup of the "ICES" participants in this dialogue should also be interpreted widely.

When considered the detailed guidance the SGMAS provides for evaluating management strategies, WGECO concurs with the points "In the simulation of a HCR, the parameterization needs to be fully documented and verified as far a possible." We further endorse their list of the major features of the target species that require consideration. Specifically SGMAS Section 4.4.2C) directs that evaluators should ask:

Does the biological part of the operating model represent the stock with a full range of plausible dynamics with respect to:

- recruitment;
- natural mortality;
- growth;
- maturity;

And at a more complex level:

- several species;
- multi-species interactions;
- cannibalism
- spatial aspects;
- seasonal/temporal aspects;
- density dependence;
- length based dependence;
- covariance between variables; and
- auto-correlation in, for example, recruitment.

As developed in Section 5.2, this list provides all the opportunities necessary to address the effects of the environment on the target species of the fisheries. As such is a major positive feature of the progress made by SGMAS in preparing ICES to work within a management strategies framework.

Likewise, in Section 4.4.2.E SGMAS lists the considerations that need to be addressed explicitly in the simulation testing of management strategies. From this long list of considerations, WGECO welcomes the inclusion of robustness to uncertainty about future states of nature as one of the key issues to address in the evaluation. In looking for how SGMAS proposes that this be done, we take particular note of the direction by SGMAS to investigate:

- How sensitive is the HCR to assumptions (e.g. recruitment model)?
- Are there exceptional circumstances that need to be kept in mind, such as shifts in regime or change in state of stock outside the current data range that will require revaluation of the management strategy?

- State a time period or duration after which certain elements should be verified or evaluated.
- Are there parameters of the management strategy that may need to be revised under given circumstances?
- Is there asymmetry in the errors or costs; i.e. Are there some risks that need to be avoided more than others?
- Is forgone yield a suitable measure of cost of failure?
- Are there conflicting objectives and information on trade off required between them? Does the evaluation inform on these tradeoffs?
- Can we highlight where tradeoffs between conflicting objectives seem counterproductive?
- Where short-term gains are giving major long-term losses.
- In a dialog process we can advise on questions that may be more informative than those posed at the start of the study.

The first four of these questions quoted from the SGMAS report highlight specific opportunities to consider uncertainty about future states of nature in the simulations evaluating the management strategies. These are far from all the ways that environmental forcing can make the future uncertainty, but parallel questions for other aspects of environmental forcing can be added to the list as needed. The important point is that the proposed approach explicitly accommodates these considerations, and is easily expanded to accommodate more if needed.

The remaining eight questions were all posed in the context of trade-offs and choices, but explicitly or implicitly between short-term benefits and long-term costs of the fishery, or between various types of objectives for stock status and uses of the stock that could not be achieved simultaneously. All of these questions about trade-offs and choices apply at least as much to trade-offs and choices regarding ensuring an acceptable state for ecosystem properties vs continued or increased use of target species of fisheries. These choices must be central to the ICES dialogue with clients of the advice, and fully structured into the evaluation frameworks ICES is developing. The SGMAS discussions were generally positive with regard to the ability of the overall framework for evaluating robustness of the HCRs to inform the choices inherent in these latter questions. However, again serious thought needs to be given to the question of whether a similar optimism applies when the robustness evaluations as done in the context of “ecosystem” choices. With operational ecosystem objectives still at a developmental stage, ICES (and the larger science community) has not yet conducted the necessary rigorous tests of whether evaluation approaches that are robust to uncertainties about the target species, and informative to choices about social, economic, and biological aspects of the target species, will be equally robust and informative with regard to ecosystem choices. We should make such testing a priority.

5.3.4 Case Histories in SGMAS

Lacking systematic testing of the robustness and utility of existing evaluation methods in the context of ecosystem approaches to management of fisheries, there may be some insights to be gained from reviewing how ecosystem issues were addressed in the fifteen case histories reported in section 5 of the SGMAS report. Table 5.3.4.1 below presented the results of WGECO examining each case history with regard to if and how environmental forcing and/or ecosystem effects of the fishery were considered in the evaluation of the harvest control rule or management strategy.

Table 5.3.4.1. Summary of the treatment of ecosystem considerations in four key aspects of the development and evaluation of management strategies and harvest control rules.

EFFECT OF: / STOCK	Ecosystem Consideration In Set of Objectives	Environmental Factors in Harvest Control Rules	Environmental Forcing Explicit In Operating Model	Environment Effects Explicit in Robustness Testing	Comments
Southern Hake	No	No	No	No	
Northern Hake	No	No	No	No	
Norwegian Spring-Spawning Herring	No	No	No	No	
Blackwater Herring	No	No	Yes	No	Temperature effect on recruitment
North-east Arctic Cod	No	No	Indirect	No	Weight of spawners considered but outside model
Icelandic Cod	No	Indirect	Yes	No	OM includes shrimp and capelin dynamically. HCR has a non-dynamic switch.
North Sea Herring	No	No	No	No	Does consider two recruitment “regimes”
West of Scotland Herring	No	No	No	No	
NSRAC flatfish	No	No	Indirect	No	Choose from “favourable” or unfavourable S-R functions; also multiple targets
Irish Sea Cod***	No	No	No	No	
Bay of Biscay Sole	No	No	No	No	
Western Horse Mackerel	No	No	No	No	
North Sea Sand Eel	Indirect	No	No	No	Overall management strategy assumes high predation
Bay of Biscay Anchovy	No	No	Indirectly	Under development	Assumption about recruitment outside OM

Clearly, although the potential to place ecosystem considerations into all four key components of the management strategy evaluation exists, little is being done to realize that potential. This is very much like the findings in Table 4.1 of the WGRED report (ICES 2006c), which found comparably little uptake of ecosystem considerations in the analytical methods used by the assessment working groups.

5.3.5 Standards for Simulations

This section of the SGMAS Report brings into focus a key challenge for ICES, as it plots a course for including ecosystem considerations in the developing management strategies framework for provision of fisheries management advice. Quoting from their Section 7.2.1:

“The evaluation framework will be used to perform experiments, the outcomes of which rely critically on the underlying hypotheses about this true system contained within the operating model. These hypotheses should therefore be considered carefully, and should either be conditioned on available data or have a strong theoretical basis or justification. In addition, the choice of assumptions underlying the state of the system that is created by the operating model will usually pre-determine many of the results of the simulation. Therefore, as in any experimental set-up, the set of assumptions (implicit or explicit) employed needs to be kept in mind when drawing any conclusions.”

WGECO agrees fully in the first, third and fourth sentences of the quotation, and has tried to apply them rigorously in all its work. The second sentence, though, underscores a fundamental challenge inherent in adopting an ecosystem approach to advising on the management of fisheries – or any other human activity. As underscored in Sections 3 and 7 of this report, the science community can expect to be data limited with regard to many ecosystem components now and into the future. Making the hypotheses conditioned on data will require both creative ways to use such data as do exist, and an acknowledgement that this option simply will not be available for many ecosystem considerations; some potentially important. Moreover, for many hypotheses about environmental effects on fish populations and communities, and about ecosystem effects of fishing, there remains substantial debate among experts on many theoretical points (see discussions in Rice 2005).

If ICES is not prepared to entertain seriously any hypotheses about ecosystem effects of fisheries or environmental forcing on stocks until the theoretical basis for the hypothesis is “strong”, then progress on implementing an ecosystem approach to fisheries advice will progress very slowly. However, such an approach is inconsistent with well established risk management practices followed in essentially all other environmental sciences. It is well established that that if a hypothesis of such relationships can be shown to be plausible, and either is not refutable with existing information or the information has low statistical power to test if the relationship is in fact present, then in should be considered for evaluation within these management strategies. Both risk management generally and particularly the application of precaution does not require waiting for a risk to be demonstrated with high statistical certainty before it is necessary to at least consider it in exploring options and testing robustness. WGECO acknowledges that this door to considering “plausible” alternative hypotheses about ecosystem processes within the Operating Model needs to opened very carefully, but it does need to be opened. Moreover, decades of experience with environmental risk management and more recently use of Strategies Environmental Assessments show how the door can be opened with adequate care. As reviewed last year, most fisheries data sets have very low the statistical power to test hypotheses about environmental effects on fish populations (ICES 2005a, Nicholson and Jennings 2004). In light of that, simply holding rigidly to a view that Operating Models should assume no environmental relationships until they are demonstrated beyond reasonable doubt seems incompatible with the Precautionary Approach, at the very least.

In that spirit, WGECO does call attention to the point made by SGMAS in their Section 7.2.5 on Stochasticity, that “there are several ways of introducing stochasticity. Three options are to draw from theoretical statistical distributions, to use bootstrapped model output, or to draw randomly from historical values. … Important points to consider include … trends or cyclical variations, for example in recruitment.” This type of thinking is readily amenable with making the best use possible of whatever information can be made available of environmental influences on the stock and ecosystem being exploited. Combined with a cautiously open mind to hypotheses about relationships that may be important to the stocks, this does provide a way forward for bringing these types of ecosystem considerations into the management strategy framework.

With regard to considering the ecosystem effects of fishing within this framework, other points raised by SGMAS also give cause for some optimism. All of their Section 7.2.2.4, on decision-making, is presented in the context of complexities of making decisions about the species being exploited directly. The acknowledgement that there could be a hierarchy of decision-rules, with increasingly complex ones implemented as the circumstances warranted, at least provides an opportunity to bring ecosystem effects into the formal decision process, if only as second and third-order rules. At least they would be incorporated directly into the framework, and once there explicitly, further progress could be made on making the entire system function more efficiently.

5.4 SGMAS’s Response to WGECO

Section 10 of the SGMAS Report is a direct response to the points made in the 2005 report of WGECO. It contains a number of constructive proposals, and a number of their observations carry the necessary internal dialogue within ICES further, with regard to making ICES fisheries advice actually be consistent with an ecosystem approach. This section intends to further that dialogue, although WGECO notes that SGMAS considers its job to be complete and recommends its own termination. These comments from WGECO are the next step in a dialogue that must continue with whatever parts of ICES carry on the work of SGMAS.

SGMAS notes that “We see two main pieces of work in the WGECO with respect to management strategies:

• Issues related to the role of ecosystem aspects in management strategies (e.g. how ecosystems affects fish stocks, how fisheries affect ecosystems, intrinsic value of ecosystem components).

• Issues related to the process of setting up management strategies (e.g. stakeholder involvement, adaptive management)”

This is an accurate interpretation of the WGECO messages, and the receptivity of SGMAS to them is welcome. With regard to what SMAS says it intends to do in response to those messages, SGMAS says, “At present, much of the practical work that is ongoing within SGMAS is directed towards methods for simulating the effects of harvest control rules. Ecosystem factors are presently incorporated into simulations by robustness testing: …. Ecosystem factors could also be added directly to simulations of harvest control rules if a hypothesis of the relationships is available. In the absence of such a hypothesis, ecosystem aspects could still be incorporated as unpredictable switch factors that suddenly change the relationships between some of the model components.”.

This response echoes the types of linkages that have outlines in Section 5.2 of our current report. WGECO wants to stress to ICES that the experts in SGMAS and WGECO both see the same roles and opportunities for considering the ecosystem in management strategies, and in their evaluation. Moreover, SGMAS acknowledges that despite the broad relevance of these ecosystem considerations, few and small steps have been taken to address them. It notes that "A first step of including biological interaction in the evaluation of HCRs are taken by the ICES multi-species assessment study groups …dealing with estimation of fish predation and do as such just cover a small part the ecosystem [emphasis added]. The multi-species groups have shown that the performance of the single species HCRs is often very different when evaluated in a single species or multispecies model (4M-HCR, see Section 8.2.1 of SGMAS report, ICES 2005b)." It also notes a few other cases where biological interactions among species are included at least implicitly in management strategies or harvest control rules, but notes "It would be very useful to expand both the ecosystem knowledge and the ecosystem implications in simulations of harvest control rules. … It would be possible to include these reservations for other species [sand eel for seabird feeding] more explicitly." This accords with the messages emerging from WGECO's review of the rest of the SGMAS report. Most of our ecosystem concerns fit readily within the proposed approaches to development, evaluation, and use of management strategies. However, there are few actual attempts to explore this potential.

SGMAS points to the right places to conduct such explorations, stating "Input from WGECO on parameterization of any effects / influences for use in predictions into the future would be of great assistance. Such aspects relating more explicitly to parameterizing ecosystem aspects/services would be especially beneficial. The aim of evaluating harvest control rules is that ICES can provide feedback on a tactical component of the fishery system. These evaluations should take the environmental conditions into account under which they operate." Both Expert Groups agree on *what* to do. There are undoubtedly some limitations on progress because we don't know everything about *how* to do some of these tasks. […/…] Questions on representation of ecosystem relationships (incoming or outgoing) to stock and fisheries, when both the data are incomplete and the parametric form of the functional relationships may not be known, must become a priority for ICES. Without focused research effort, little progress can be expected. Moreover, WGECO stresses that there is an issue of assessment and advisory culture to be confronted as well. This cultural issue is discussed in Section 5.2.5 of this Report, and needs to be discussed seriously among a properly diverse group of disciplinary experts within ICES, and then with clients of ICES advice.

In addition to the common view of the appropriateness and tractability of forming direct links to the ecosystem in both the incoming (environmental forcing on stock dynamics) and outgoing (ecosystem impacts of fishing) parts of the overall management strategy process, SGMAS and WGECO agree on a number of process and governance issues. Neither SGMAS nor WGECO has developed these aspects of their work to the point where specific recommendations have been made to either ICES or clients of ICES advice. However the common acknowledgement by both groups of the need for action in these areas should be taken up by the Advisory and Science processes of ICES, and become central to dialogue with its clients. In particular WGECO notes that SGMAS agrees:

> "There is some resemblance between the SGMAS guidelines for evaluation and the Strategic Environmental Assessment (SEA) of fisheries that is described by WGECO. There is scope for integrating these two approaches to make sure that all the relevant aspects are covered at an operational level to give an overall evaluation framework; one important area for joint development is how to allocate / interpret the different components of such a scoring process to arrive at meaningful and reliable advice in the event that there are conflicting signals."

It will be essential for ICES to address this issue in the near term , if it is to provide advice in an integrated ecosystem context. The need for a comprehensive framework in which both fisheries and ecosystem considerations are addressed in a single evaluation, and in which "conflicting signals" can be expected often, must be met before provision of integrated advice will be possibly in even a semi-quantitative way. and

> "The involvement of different parties in the definition of management strategies and tactical decision making is a very common feature [internationally]. This is now also captured in our description of the management strategy where the questions of who is participating and how are they participating are prominent features. SGMAS is not aware of evaluations of such arrangements. However, this is important food for thought."

WGECO agrees that this is important food for thought. Such an evaluation would require a good integration of social and biological scientists, but ICES is well placed to promote such linkages and foster such an evaluation.

Finally WGECO notes the proposal by SGMAS that "it would be better to actively incorporate the expertise of WGECO in groups that are focussed on carrying out evaluations of actual management strategies or harvest control rules. Such an approach would provide a strong incentive to integrate ecosystem aspects in management strategy evaluations in a concrete way." We agree that is desirable and possibly mandatory, if progress is to be made. The clear message in both Table 5.3.4.1 of this report and Table 4.5.1 of WGRED (ICES 2006c) is that assessment working groups as they are currently constituted are showing little uptake of these issues. Unfortunately, experts knowledgeable in both assessment methodology and the fisheries management aspects of detailed ecosystem issues are in even shorter supply that experts in either field alone. Genuine capacity building in this intersection of expertise is essential if progress is to be made.

5.5 Conclusions and Recommendations from Sections 5.3 and 5.4

WGECO notes that the situation represented by Table 5.3.4.1 is tacitly acknowledged by SGMAS in its reply to WGECO. The *status quo* is unacceptable to WGECO and should be unacceptable to ICES. The table documents clearly the need for cultural change and action within ICES.

4) In any guidance documents prepared for use by Expert Groups developing or evaluating management strategies or harvest control rules, explicit consideration of both environmental forcing and ecosystem effects of fisheries should be treated as a core part of the work; it should be "business as usual" not an optional afterthought.

5) Ecosystem experts should be included in all teams developing and evaluating management strategies and harvest control rules. This will present an even greater capacity challenge than finding sufficient fisheries experts for such teams, but the challenge should highlight where capacity building in member states is most needed.

6) The "knowledge development system "for any management strategy initiative should implement fully recommendations 1, 3, and 4 from Section 5.2.3 of this report.

7) ICES should conduct some tests of the appropriateness of the "classic 3-stage model" for use with ecosystem indicators suitable for management decision-making, instead of SSB as the x-axis. WGECO would be one Expert Group capable of conducting such tests.

8) ICES should conduct some tests of how a hierarchy of decision rules (as proposed in SGMAS Section 7.2.24) would function, when the decision rules took into account both fishery and ecosystem objectives. WGECO would lead such tests, but would require additional fisheries expertise as well.

9) ICES should identify a couple of suitable test cases for simulation and robustness testing in the face of choices and trade-offs of ecosystem and fisheries objectives, as per the questions in Section 5.3.3 of this report and 4.4.2 of SGMAS, and evaluate how well those types of choices fit within the frameworks being developed for evaluation options concerned solely with uses of the target species

in the short, medium, and long term. WGECO experts should participate in such evaluations, but they would probably be best led by experts in management strategy evaluations.

10) If SGMAS continues, or another Expert Group is created to carry this work forward, WGECO reiterates its call from 2005 that a joint or overlapping meeting would be beneficial to both groups.

11) Perhaps most importantly, ICES must commence a change in culture and mind-set from an attitude that environmental considerations only need to be addressed and included in management strategy evaluations when there is compelling empirical or theoretical documentation that a relationship or impact exists (see Section 5.4), to an attitude consistent with good risk management practices applied in many other fields. That is, when scientifically a plausible case can be made that a serious risk may be present, and the statistical power to test for presence of the relationship or impact is low with existing data, then the risk must be considered in the representations being considered for system dynamics, the options being developed for management choices and in the testing of robustness of options to failure.

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6 Development of EcoQO on changes in the proportion of large fish and evaluation of size-based indicators

d) Review and report on the work of WGFE 2006 in their further development of the EcoQO on changes in the proportions of large fish and hence the average weight and average maximum length of the fish community, and complete the evaluation of the utility of size based indicators in management frameworks.

6.1 Introduction

WGECO has advised on the development and implementation of the Ecological Quality Objective (EcoQO) approach for many years (ICES 2001; 2002; 2003a; 2004a; 2005a). The consequences of the work done by WGECO are reflected clearly in the Bergen Declaration, adopted in March 2002. An extensive overview of the scientific work done by ICES to underpin the EcoQO framework, as well as its application in ecosystem management, is given in ICES (2005c).

At the 2006 meeting of the OSPAR Biodiversity Committee (BDC) an overview of revised EcoQO system for the North Sea was presented by the OSPAR secretariat, based on the conclusions adopted by OSPAR 2005 on the North Sea Pilot Project. The EcoQ on fish length has not been included in the list of 12 EcoQOs as it has not progressed far enough, and the work presented in this chapter is intended to provide enough guidance for ACE to advise OSPAR **conclusively** on this EcoQ.

The development of this EcoQO has been well-studied by ICES and its' relevance as an objective for fish communities and the linkage with fisheries make it suitable for use in the EcoQO framework. However, this does not preclude that ICES has previously identified weaknesses in the EcoQO framework (REFS).

6.2 Review of progress made in WGFE

The Working Group on Fish Ecology (WGFE) has been studying the Ecological Quality Element (EcoQ) on fish length for several years (ICES 2003b; 2004b; 2005b; and 2006). In 2003, 2004 and 2005 the group carried out case studies in order to identify suitable size-based indicators. In 2005 they also summarised the indicators they had identified. At their most recent meeting the group attempted to synthesise the available knowledge by: (1) identifying a range of ecological objectives which could potentially be informed or monitored by a large fish index (which might include the proportion of large fishes); (2) summarising the different measures of large fishes; (3) mapping large fish measures against the appropriate objective; and (4) evaluating each measure against indicator selection criteria as described by Rice and Rochet (2005) (ICES, 2006).

The **objectives** described by the group were related to: large fish as large predators (size, trophic structure and predatory function); assemblage reproductive capacity; conservation of threatened and declining species; wider biodiversity; and charismatic species. Although this is a thorough overview of the possible ecological objectives, WGFE themselves realised that the latter three objectives were unlikely to be well represented by an indicator of size and other objectives were more suited. The assemblage reproductive capacity, species-specific large fish criteria which could represent the species-specific life-history, is a promising approach which deserves more work done on it.

The **indicators** themselves were described according to a definition of "large" fish, whether they could be applied to species or assemblages, and the additional data requirements that would be necessary to implement them. For proportion of large fish three definitions of large fish were used: percentiles; arbitrary cut-offs (e.g. 20, 30, 40 cm); or biologically relevant cut-offs (e.g. length-at-maturity).

The **proportion of large fish**, either measured as a percentile or using an arbitrary cut-off, was the only indicator that scored high for all selection criteria. The criteria on sensitivity, specificity and responsiveness were not scored by WGFE, but WGECO feels that it is possible to score them as follows:

Property	Description	Applicable to size indicators
Sensitivity	Trends in the Indicator should be sensitive to changes in the ecosystem state, pressure or response that the indicator is intended to measure	Yes
Specificity	Indicators should respond to the properties they are intended to measure rather than to other factors and/ or it should be possible to disentangle the effects of other factors from the observed response.	Not predictable how one single indicator will respond, but disentangling possible
Responsiveness	Indicators should be responsive to effective management action and provide rapid and reliable feedback on the consequences of management actions.	Not readily quantifiable and not rapid

WGFE makes some proposals as to how to develop reference levels (defined as “the level of the EcoQ where the anthropogenic influence on the ecological system is minimal.”). WGECO takes these proposals and develops them further in section 6.4.2.

Overall, WGFE felt that*: “(i) the management objectives have to be clearly set so that they can be tightly linked to appropriate “large fish”-derived indicators (ii) once the objectives are set, there is a strong need to assess the sensitivity, responsiveness and specificity of those indicators.”*

Over the past years, WGFE has provided a thorough and valuable discussion of the various theoretical aspects of fish length in relation to the EcoQO, but they have not been able to suggest a definite formulation for the EcoQO, for the many reasons stated above. However, WGECO feels that the indicator proportion of large fish has been shown to be a good indicator of size-structure and will develop it accordingly.

6.3 An evaluation of the utility of size-based indicators in management frameworks

In the past few years there have been numerous studies on size-based indicators and their use in management frameworks, many of them presented at a Symposium on Quantitative Ecosystem Indicators for Fisheries Management which was held in Paris in 2004. The proceedings are published in the ICES Journal of Marine Science Vol. 62(3). There are a number of articles exploring the utility of size-based indicators (Shin *et al*., 2005; Jennings and Dulvy, 2005; Rochet and Rice, 2005). In Chapter 3.7.2.1 of this report there is a discussion of a number of different types of indicators to reflect trophic relationships.

Shin *et al*. (2005) carried out an extensive review of the use of size-based indicators (SBI) to evaluate the ecosystem effects of fishing and concluded that: *“SBIs are sensitive to variations in fishing intensity. Reference directions of change can be established on the basis of theoretical, empirical, and modelling studies. In some cases, response time may be improved by suitable selection of the most informative size classes, and by improving survey design (increased standardization and replication within strata). Although a slow response to changes in exploitation limits their use in the context of short-term, tactical fisheries management, the failure of conventional management systems to sustain fisheries has led to a strong movement towards strategic (5–10 year) approaches to managing fisheries*

(Butterworth and Punt, 1999; Geromont et al., 1999; Smith et al., 1999). In this context, SBIs score high for inclusion in the suite of indicators required for an ecosystem approach to fisheries (EAF).”

Furthermore: *“no single SBI can serve as an effective overall indicator of heavy fishing pressure. Rather, suites of SBI should be selected and reference directions may be more useful than reference points. Further modeling and worldwide comparative studies are needed to provide better understanding of SBIs and the factors affecting them. The slow response to fishing pressure reflects the complexity of community interactions and ecosystem responses, and prohibits their application in the context of short-term (annual) tactical fisheries management. However, movement towards longer-term (5–10 years) strategic management in an ecosystem approach to fisheries (EAF) should facilitate their use.”* (Shin *et al*., 2005)

In their study, Jennings and Dulvy (2005) stated that *“Practical issues preclude the development and adoption of firm reference points for size-based indicators. However, an appropriate target to support ecosystem approach to fisheries management (EAFM) would be a reference direction that is consistent with a decline in the overall human impact of fishing on the community, and thereby on the ecosystem.”*

Piet and Jennings (2005) showed that although the indicators for slope of biomass-size spectra, mean weight and mean maximum length showed broadly consistent responses to fishing effort, only the slope of biomass-size spectra showed a response to the spatial management measures carried out at traditional time and spatial scales.

Daan *et al.* (2005) have explored size and L_{max}-spectra to identify the indirect effects of fishing on the North Sea fish community. They showed, based on trawl surveys, that the abundance of small fish and the abundance of fish with a low L_{max} have steadily and significantly increased during the past 30 years. They identified a time lag from the time at which fishing effort was highest in the mid-80s to the present day, which supports the earlier conclusions made on responsiveness of SBI for management.

Greenstreet and Rogers (2006) analysed Scottish groundfish data from 1925-1997 in order to identify potential reference levels for an ecosystem approach to management. The authors consider a variety of different indicators of the fish community – size composition metrics, life history characteristic metrics, species richness and diversity metrics and trophic level metrics, as well as their interactions. For the metrics ‘percentage of large fish’, ‘average fish weight’ and ‘average L_{inf} of the community’, the authors demonstrated a definitive effect of fishing and they suggest potential reference levels, specific to the Aberdeen 48 ft trawl gear and for the NW North Sea, of 10%, 125 g and 48 cm, respectively.

In their recent review, Shin et al (2005) suggest that the use of size-based indicators within a management framework aimed at mitigating the effects of fishing on the broader fish community has some drawbacks. Like many others, these indicators are not specific to fishing. Rather, environmental and density dependent effects on growth and recruitment rates may also affect metrics of fish size regardless of fishing activity levels (Ricker 1995; Ottersen and Loeng 2000; Lekve et al 2002). Poor recruitment may cause the average size of fish in a community to increase as populations become progressively more dominated by older individuals (Wilderbuer et al 2002). Conversely, increased rates of recruitment may cause the mean size of fish to decline even in the absence of over-exploitation by fisheries. In situations where over-exploitation has been followed by remedial action (eg fishery closure), coincidental increases in recruitment rate may delay the anticipated increase in size-based metric values (Badalamenti et al 2002). These biological considerations make size-based indicators often insensitive on annual or near-annual time scales. However on time scales appropriate for applying indicators of ecosystem health (usually half-decadal or more) only persistent periods of recruitment failure, steady increases in recruitment strength, or similar persistent changes in growth rate, would render these size-based indicators uninformative about changes in fishing mortality that had been implemented during the interval. The same monitoring necessary to produce any of the size-based indicators could be expected to inform analysts about any such persistent changes in recruitment or growth rates.

Fishing induced changes in the life-history trait composition of fish communities may also need to taken into account when considering the use of size-based metrics as indicators. Stoberup et al (2005) suggest that metrics of fish community size structure may be poor indicators of over-exploitation in fish communities characterised by fast growth rates, small body size and early age at maturation. Thus, because of the well documented changes in community life-history character composition caused by fishing (Jennings et al 1999; Piet and Jennings 2005; Greenstreet and Rogers 2006), as communities become dominated by species with small (perhaps < 30 cm) L_{max} , metrics of fish community size structure may become less effective as indicators of short-term responses to reduced fishing. Species with relatively larger L_{max} need to become re-established in the community before the size-based indicators will reflect improving ecological status for the community. However, continued over-fishing of communities dominated by species with small L_{max} will still lead to further reduction of size-based indicators.

This discussion serves to make the point that, despite their apparent advantages, metrics of size in fish communities should still be used with care. The potential for processes other than fishing mortality to influence trends in metric value needs always to be considered. Size-based indicators are likely to perform most weakly in situations where over-fishing has been chronic for some time. Under such circumstances, debate about the need for action ought not depend critically on the current values of the indicators in question. It is also likely to take longer to detect improvements in indicator values following the implementation of remedial management action (Nicholson and Jennings 2005) then if appropriate action had been taken before the community became severely altered by fishing. Under such circumstances, because of the extent of improvement required, remedial action is likely to take longer anyway.

In most studies where metrics of size in fish communities have been applied, the anticipated results have been observed (eg Zwanenburg 2000; Bianchi et al 2000; Piet and Jennings 2005; Daan et al 2005; Blanchard et al 2005). Blanchard et al (2005) consider possible confounding effects caused by environmental variation, but conclude that fishing had the stronger effect on community size structure. Greenstreet and Rogers (2006) conclude that variation in fishing effort was the principal cause of differences in the size structure of the groundfish assemblage.

6.3.1 Concluding review sentences

All the work done on size-based indicators shows that they are affected directly and indirectly by fishing. However, making the EcoQO operational as a management tool, with performance indicators, metrics, and associated reference points is far from straightforward, as is clear from the above "review". This may have contributed to the lack of adoption of this EcoQ by OSPAR for use in management contexts. A number of suggestions have been made for the use of size-based indicators in an ecosystem approach to fisheries management, including using suites of indicators and identifying reference directions. Building on these suggestions and acknowledging the difficulties and uncertainties discussed above, WGECO proposes a way forward, based on work done by WGFE and elsewhere.

6.4 WGECO advice on implementing the EcoQO

6.4.1 Background

The EcoQ element on changes in the proportions of large fish and hence the average weight and average maximum length of the fish community (as measured in research trawls) is clearly linked to fishing. As such it is a useful state indicator and a good measure of this component of ecosystem health. Although these metrics clearly serve as useful indicators of the effect of fishing on the whole fish community, ICES has advised that they are also indicative of wider changes in the ecology of the fish community, and biodiversity of the ecosystem. Reduction in the mean size of fish in the community has implications for trophic structure (fewer large fish, more small fish), for reproductive potential of the larger predators in the community, and the overall role of fish in ecosystem functionality.

The EcoQ element reflects two interrelated aspects of fish communities:

- size-structure: proportion of large fish and average weight
- species composition: average maximum length of the fish community

The proportion of large fish and the average weight are, therefore, applicable to the size-structure of the fish community. The measure of average maximum length of the fish community is related to the distribution of species with specific life-history characteristics. Both indicators can be considered complimentary when using them to assess the effects of fishing on the fish community.

6.4.2 Objectives and implementation

WGECO concludes that the EcoQO can be further progressed as part of an objectives-based management framework, and so has defined an overarching objective, consistent with high level international agreements (e.g. World Summit on Sustainable Development) to achieve 'by 2010 a significant reduction in the current rate of biodiversity loss'.

WGECO therefore suggests the goal for the fish community to be:

Halt as rapidly as possible, and begin to reverse by 2010, both the decline in the mean weight and the proportion of large fish.

The setting of reference levels, both in the sense of OSPAR "reference level" (the level of the EcoQ where the anthropogenic influence on the ecological system is minimal."), and in the way that ICES uses reference points in scientific advice could not be done at this meeting, and may prove challenging because of constraints of data availability. Although long-term international monitoring programs exist in the North Sea (e.g. SAGFS, IBTS), changes over time in the gears used, sampling practices such as haul duration, and indeed the size and power of vessels employed, make the survey data inconsistent over the periods involved. The data from these monitoring programs will have to be made as comparable as possible before any reference points for use in advice can be estimated. Furthermore, even the longest time series of data available started long after the time at which fishing may have affected the fish community. This poses particular problems for the estimation of "reference levels" as defined by OSPAR.

Regarding the two metrics, mean weight of fish in survey samples is readily defined. However, the proportion of large fish in the community needs "large fish" to be defined. In a recent analysis of long-term data for the northwestern North Sea, Greenstreet and Rogers (2006) defined large fish as those above 95% of the cumulative frequency distribution of all fish lengths sampled over the entire 1925 to 1996 period. Only 5% of all the individuals caught exceeded 30cm. They therefore defined large fish as fish of greater than 30cm in length.

These data also provide a basis for estimating reference levels. They suggest that, between 1925 and 1997, the proportion of large fish in heavily fished areas has declined by more than a factor of 3 from the levels in areas where fishing activity has been negligible, and the mean weight of fish has decreased by 50% (Greenstreet and Rogers 2006). Although this time series is long enough to show the fishing induced changes in these metrics, it was discontinued in 1997. Consequently, it is not possible to set precise metric reference levels for management based directly on these data. However, the data do suggest the relative changes that could be applied to current and future survey data. Thus, to achieve the OSPAR definition of reference level would require at least a tripling of the proportion of fish over 30 cm and a doubling of the mean weight of fish, relative to the 1997 values in any groundfish survey used for monitoring in the North Sea (e.g. the Q3 IBTS survey).

OSPAR is clear that in its objectives-based management approach, management targets are not intended to be the pristine "reference level". Such objectives would preclude the continuation of sustainable human activities. Rather management targets should be guided by societal choices regarding the costs and benefits of moving towards such reference levels, and reflect the overall commitment in WSSD and the CBD to conservation and sustainable use of ecosystems. Thus, one possible approach would be to look back in time to determine a period when fishing was considered sustainable. Early in the 1980s was the last period when ICES advice regarding the management of the exploited species was generally for the maintenance of "*status quo*" exploitation rates, suggesting that this was the last period when science experts considered fishing to be generally sustainable in the North Sea. Although the data in Greenstreet and Rogers (2006) suggests that, by the early 1980's, both metrics had clearly departed from the "non-fished" OSPAR reference level. Nevertheless, the proportion of fish over 30cm was 1.4 times higher, and the mean weight of fish was 1.3 times higher, than current values, and ICES advice was that fishing levels were sustainable.

In the short-term therefore, operational targets for both metrics could be:

1. **Halt the decline in the proportion of fish greater than 30cm in length in survey catches immediately.**
2. **Halt the decline in the mean weight of fish in survey estimates immediately.**

In the medium term, a suggested approach would be as follows. First review ICES advice to establish the fishing mortalities considered sustainable in the early 1980s (translating those F's onto scales consistent with current assessment practices). Next, using reasonable assumptions about stock productivities from recent assessments, estimate how many years [year "x"] it would take for stocks to rebuild to have the proportion of fish greater than 30cm in length to 1.4 times 1997 survey estimates, and how many years [year "y"] it would take for the mean weight of fish to rebuild to 1.3 times the 1997 survey estimates. Then the medium term operational objectives would become

3. **Restore the proportion of fish greater than 30cm in length to 1.4 times 1997 survey estimates by [year "x"].**
4. **Restore the mean weight of fish to 1.3 times 1997 survey estimates by [year "y"].**

Such operational targets would serve to reverse the current negative trends, and should be consistent with the continuation of the fishing industry at a sustainable level.

In the longer term, these targets could be revised taking account of improved information on the ecological consequences of an over-fished fish community, societal choices for more or less ambitious conservation objectives, and the consequences of achieving these medium-term targets on the fishing industry.

In summary, for the development of this EcoQO, WGECO can suggest the following stepwise approach:

- **Short-term**. Aim to cease the decline in proportion of large fish as described above by lowering total fishing mortality (taking into account discarding, etc). Existing

ICES advice to reduce F on many target species of fisheries will contribute to this objective, but additional management measures may be necessary and will speed achievement of this conservation goal.

- **Medium-term**. Establish operational reference points according to the strategy outlined above to restore the proportion of large fish and the mean weight of fish, and implement management plans with appropriate control measures to facilitate their achievement.
- **Long-term.** Continue to improve the scientific basis of our estimates of reference points for proportion of large fish and mean weight indicators. Because of the limitations on historic data, empirical analysis of them will have to be augmented by ecological modelling studies to provide insight into:
    - fishing mortality rates for the fish community at which there are no serious biodiversity concerns,
    - the proportion of large fish that is associated with such mortality rates
    - how both the sustainable mortalities and proportion of large fish might change over time.
- In addition, discuss with policy (EMS, EU, OSPAR) states of the fish community and corresponding target levels for proportion of large fish that are acceptable to society as targets. Using the modelling approaches above, determine how long it should take to move the indicators to the target levels. The modelling will inform the setting of realistic time-frame for both the achievement of targets and our ability to measure progress towards them.

6.4.3 Finally

WGECO considers that this EcoQO has been rigorously tested and is ready for implementation. Accordingly WGECO proposes that OSPAR consider the objective and indicators described above and determine how to proceed in the implementation of this EcoQO.

Suggested ToRs

For the Working Group on Fish Ecology (WGFE)

I. Review current stock assessments to determine reasonable assumptions for the productivity (recruitment, growth and natural mortality) of fish stocks in their current state for as many stocks as possible in the North Sea.

II. Using these estimates of productivity for assessed stocks, combined with reasonable assumptions from life-history theory for non-assessed species, estimate future trajectories of the two fish community size-based indicators (proportion of fish >30cm and mean weight of fish) under a variety of different fishing mortality (*F*) scenarios determined across the fish community to which the indicators is applied (ie the part of the community sampled by the specific research surveys).

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

I. Review the work of WGFE and consider the results of the analyses carried out.

II. Use the results of the analyses to complete the work implicit in section 6.4.2 of WGECO’s 2006 report to complete the development of fully quantified and operational EcoQOs for the two size-based indicators. This should include:

a. Fully quantified targets for the two indicators

b. Estimation of projected time-scales by which these targets might be achieved under a variety of different remedial fisheries management strategies (reductions in "community" *F*).

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7 ICES capacity to advise the Regional Advisory Councils on the ecosystem effects of fishing

ToR 'e' *For each area for which a Regional Advisory Council is established, or is under development, review the preparedness of ICES to advise on the ecosystem effects of the fisheries relevant to the RAC. Where deficiencies are identified, consider the risks posed by the gaps, and suggest feasible steps to redress the gaps in the short or medium term.*

7.1 Background

ICES have begun a dialogue with the RACs to develop an understanding of their requirements for advice and how this advice might be provided. This process began following the meeting of MCAP in September 2005. The report of this meeting, as presented to the ICES Council in October 2005, states:

The Chair proposed that ICES will arrange a "Dialogue meeting" in 2006 bringing the RACs and ICES together to identify:

- *What the RACs may want from ICES*
- *What ICES can deliver*
- *Identifying the form for cooperation between RACs and ICES*

The MCAP chair indicated in his presentation to the ICES Council in October 2005 that clarifying the relationship between ICES and the RACs would be a high priority issue in 2006. MCAP then arranged a dialogue meeting with representatives from the RACs at ICES on 20-21 Feb 2006.

The ICES Bureau discussed RAC-ICES interactions at their meeting in January 2006 and concluded that ICES would be keen to respond to requests from RACs on scientific issues within the remit of ICES. The main issues for ICES would be the availability of funding and scientific resources. ICES would, as with most aspects of ICES work, be dependent on the (national) organisations that employ relevant scientists. These organisations would need to allow these scientists to undertake this work for ICES. Funding is also required to cover ICES Secretariat work, travel and per diem costs associated with answering any requests. RACs do not have the funding to pay for direct requests to ICES, and thus far they have been dependent directly on those institutions that are willing to supply resources and scientists.

If the RACs were to request advice from ICES then there were two routes that they could take; either to send the requests to ICES via the EC or via a Member State. Since the ICES – EC MoU is due for revision in 2006, the EC participants at the ICES-RAC dialogue meeting invited the RACs to discuss any special issues they would like to see covered in the MOUs with the Commission.

7.2 The RACs

The RACs were established by Council Decision 2004/585. Table 7.2.1 summarises the RACs, the regions and countries which they cover and background information on their status (ICES, 2006). The substructures of the RACs are not consistent in terms of sub-regions or métiers. For example, the North-western waters RAC is structured by area, with groups working on (a) the West of Scotland ICES Areas Vb (EC) and Via, (b) the western approaches/ west of Ireland and Celtic Sea ICES areas VIb, VII (except a,d,e), (c) the English Channel ICES areas VII d, e and (d) the Irish Sea ICES area VIIa. By contrast, the North Sea RAC is structured by topic and area, with groups working on (a) Demersal, (b) Flatfish, (c)

Spatial planning/ MPAs, (d) the Kattegat and Skagerrak and (e) Socioeconomics. These different sub-structures imply that advice may be requested in different ways, either relating to an area or to a specific fishery.

Table 7.2.1. The RACs, the regions and countries with interests and their status.

Regional Advisory Council	**ICES Areas covered and countries with interests in the RAC**	**Status of RAC**
Baltic Sea	IIIb, IIIc and IIId. Countries with interests are Denmark, Germany, Estonia, Latvia, Lithuania, Poland, Finland and Sweden.	The Baltic Sea RAC was established on 15 March 2006.
North Sea	IV, IIIa. Countries with coastal are Belgium, Denmark, Germany, Spain, France, The Netherlands, Poland, Sweden and the UK.	The North Sea RAC was established on 1 November 2004.
North-western waters	V (excluding Va and only EC waters in Vb), VI, VII. Countries with interests are Belgium, Spain, France, Ireland, the Netherlands, and UK.	The North Western Waters RAC was established on 26 September 2005.
South-western waters	VIII, IX and X (waters around Azores), and CECAF divisions34.1.1, 34.1.2 and 34.2.0 (waters around Madeira and the Canary Islands). Countries with interests are Spain, France, and Portugal. Belgium, the Netherlands and the UK.	Representatives of the fisheries sector have met five times since 2004. The most recent preparatory meeting took place in Brussels on 30 November 2005.
Mediterranean Sea	Maritime Waters of the Mediterranean east of 5°36’W. Countries with interests are Greece, Spain, France, Italy, Cyprus, Malta and Slovenia.	The EU members of Medisamak (Association of Mediterranean Fishing Professionals) have taken the lead in establishing the Mediterranean RAC. Two preparatory meetings have been organised so far, with the participation of shipowners only: on 1 March 2005 in Tarragona and on 12 September 2005 in Madrid.
Pelagic	All pelagic fisheries. Countries with interests are Denmark, Germany, Spain, France, Ireland, the Netherlands, Poland, Portugal, Sweden and UK.	The Pelagic RAC was established on 16 August 2005.
High seas/long-distance fleet	All areas not covered by RFOs. Countries with interests are Denmark, Germany, Estonia, Spain, France, Ireland, Italy, Latvia, Lithuania, the Netherlands, Poland, Portugal and UK.	The Spanish shipowners have taken the initiative to coordinate the establishment of the Long Distance RAC by organising several preparatory meetings. The most recent preparatory meeting took place on 25 October 2005 in Brussels.

7.3 Ecosystem effects of fisheries relevant to the RAC

Consistent with the ToR, we focus on the preparedness of ICES to address requests from the RACs that relate to the ecosystem effects of fisheries. It is assumed that requests are most likely to relate to those effects that compromise the achievement of the objectives of the CFP. The most relevant of objectives of the CFP are those that require measures to ‘limit the environmental impact of the CFP’ and refer to the ‘progressive implementation of an ecosystem-based approach to fisheries management’.

WGECO recognised that it was not realistic to undertake a detailed review of the ecosystem effects of fishing in all RAC areas, although WGECO has attempted to provide a ‘high level’ review of known fishing effects in each RAC area in relation to the components of the ecosystem. However, for the North Sea RAC, which is the longest established RAC and arguably the RAC for which the data needed to assess the effects of fishing on the ecosystem are most abundant, we have provided a relatively comprehensive assessment of the areas where ICES could advise on the impacts of different métiers on ecosystem components. This is possible due to the detailed review of interactions and fishery components that WGECO has conducted (see Section 3 of this report) and provides an example of an approach that could be adopted for other RACs as more information comes available. For all RACs, our assessment highlights data deficiencies that may affect ICES preparedness to advise on the ecosystem effects of fishing.

7.4 Ecosystem components

In 2004 WGECO developed a list (Table 7.4.1.) of key ecosystem components that could be used to guide the development of management measures aimed at delivering ecosystem level objectives.

Table 7.4.1. List of ecosystem components identified by ICES (2004).

Habitats – physical and chemical attributes
Nutrients
Plankton (phytoplankton and zooplankton)
Benthos
Fish community
Commercial fish and shellfish
Marine mammals (including marine mammals)
Seabirds

7.5 Assessing capacity to provide advice at the RAC scale

We assessed the capacity of WGECO to advise on the ecosystem effects of fishing at the RAC scale in terms of the state of knowledge of the biological effects of the fishing impact that results from the spatial and temporal distribution of fishing (knowledge), the severity of fishing effects based on the probability that they may compromise the achievement of CFP objectives (severity) and the availability of data on the spatial and temporal distribution of effort or rates of mortality attributable to a métier (data). Knowledge, severity and data were all categorised on a simple ‘high’, ‘medium’, ‘low’ and ‘unknown’ scale following Table 7.5.1.

Table 7.5.1. Categories used to describe the state of knowledge, severity and availability of fleet data when providing advice on the effects of fishing in the RACs.

Issue	Knowledge	Severity	Data	Scale
Description	State of knowledge about biological effects of the impact	Severity of fishing effects in relation to CFP objectives	Accessibility and availability of data on fleet activity	The scales at which advice on the identified impact could be given
Category 1	Detailed: knowledge is sufficient to give rigorous quantitative advice	Expected Compromise (Expected C): Known ecosystem effect of fishing that is expected to compromise the achievement of CFP objectives	Sufficient All (SA): Data accessibility and availability is sufficient to support the provision of quantitative advice for all major fishing grounds/ métiers.	Finest: Advice could be given at scales smaller than the ICES rectangle
Category 2	Sufficient: knowledge is sufficient to give informed judgement	May Compromise (May C): Fishing effects that may compromise the achievement of CFP objectives	Sufficient Some (SS): Data accessibility and availability is sufficient to support the provision of quantitative advice in some areas/ métiers at the stated scale	Rectangle: Advice could be given at the scale of the ICES rectangle
Category 3	Insufficient: knowledge is not sufficient to provide advice	Unlikely Compromise (Unlikely C) Fishing effects are known but are unlikely to compromise achievement of CFP objectives.	Insufficient: Insufficient to provide advice in most areas/ métiers	Area: Advice could be given at the scale of the ICES area
Category 4	-	Unknown Insufficient knowledge base to make a judgement about the effects of fishing	-	RAC: Advice could be given only in general terms at the scale of the RAC

The métier can be defined at a number of levels within any RAC area. In the North Sea example, broad high-level categories were identified, but these can be further subdivided as discussed in Section 6.2.2. of the 2005 WGECO report (ICES, 2005; Figure 7.5.1.). The level in the hierarchy at which management action will be taken will depend on the specificity of the impact that management is expected to address. For example, a widespread impact caused by many gear types (e.g. the impact of many mobile gears on benthic habitats) may need to be managed far more generically (e.g. by broad spatial control) than an impact associated with a specific gear type (e.g. bycatch of seabirds in a long-line fishery).

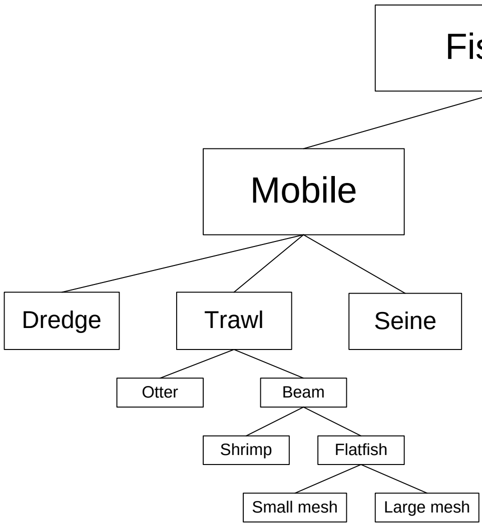

Figure 7.5.1. Example of how a fishery can be sub-divided into increasingly more defined métiers (ICES, 2005).

In general terms it will be more straightforward to give generic advice that applies to high level métiers, since this advice can often draw on generic understanding of the effects of fishing. For more defined métiers and métier-specific impacts, new and specific research is likely to be needed.

7.6 Fishing effects on ecosystem components

For the North Sea and Baltic Sea RACs, the ecosystem components identified in Table 7.4.1 were linked to the métiers identified in Section 3 of this report to provide an assessment of the state of knowledge and severity of the ecosystem effects of fishing in relation to each interaction. For other RACs the knowledge of relevant métiers in WGECO was not so extensive and example information was tabulated at the scale of the RAC with information provided on the relevant métier when available. In many cases in these other RACs, generic knowledge gained from the North Sea and in other studies may be used to provide advice.

7.6.1 The North Sea RAC

Table 7.6.1.1 shows the outcome of this analysis, the capacity of ICES to advise on each interaction and whether the knowledge base to give this advice is adequate in relation to (1) understanding of the fishing effect (2) availability of data to conduct relevant analysis.

Table 7.6.1.1. The effects of fisheries in North Sea métiers (as identified in Section 3) on ecosystem components and the preparedness of ICES to advise on those effects based on criteria for knowledge (detailed, sufficient, insufficient), severity (expected to compromise CFP objectives, may compromise CFP objectives, unlikely to compromise CFP objectives), data (sufficient all areas, sufficient some areas, insufficient) and scales at which advice on the impact could be given (finest, ICES rectangle, ICES area, RAC). Further details of criteria are in Table 7.5.1.

Effect on component	**Métier:** Issue	**Beam trawl**	**Otter trawl for fish and Nephrops**	**Dredging**	**Small mesh industrial fisheries**	**Fixed gears**	**Pelagic**
Impact on sensitive **habitats**	Knowledge	Detailed	Detailed	Detailed	Sufficient	Detailed	Sufficient
	Severity	May C	May C	Expected C	Unlikely C	Unlikely C	Unlikely C
	Data	SS	SS	Insufficient	Insufficient	SS	Insufficient
	Scale	Finest, rectangle, area, RAC	Finest, rectangle, area, RAC	Rectangle, area, RAC	Rectangle, area, RAC	Rectangle, area, RAC	Rectangle, area, RAC

Nutrients	The effects of fishing on nutrient flows and bio-geochemical rates are related to the physical disturbance by gears in contact with the seafloor. The resuspension of sediments that occus during the trawling process may be associated with the release of nutrients (and contaminants) that have previously been stabilised in the sediments. Most of the available literature on nutrient flux is not from the North Sea although work is currently being undertaken in the southern North Sea (Trimmer *et al*., 2005); most of these effects will not however be specific to the North Sea. The variance in effects between métiers is related directly to the scale and frequency with which the gear disturbs the seabed and the nature of the seabed over which the gear is being towed. Gears that penetrate deeper into softer sediments will change nutrient flux much more than those that are towed over harder sediments. Gears that do not touch the seabed will be unlikely to affect nutrient flux. Advice can be provided at the Rectangle, area and RAC levels.						
Plankton	To the best of our knowledge there are no significant effects of fishing on plankton (phytoplankton or zooplankton). While we acknowledge that change in the population size and distribution of plankton feeding members of the other components may itself be a consequence of fishing effects, there is no known evidence that this is a significant driver in the structuring of North Sea plankton. Changes in the abundance of fish and benthos, from the direct and indirect effects of fishing, will alter the total amount, and spatial distribution, of larvae produced. In many regions the seasonal input of meroplanktonic larvae comprise a major part of the zooplankton and influence system dynamics through their consumption of phytoplankton and microzooplankton. Similarly, there are certainly occasions when large, gelatinous, plankton are caught in or macerated by passage through nets. We are not aware of any studies that allow us to comment on the ecological consequences of this mortality. Advice can be provided at the Rectangle, area and RAC levels.						
Biodiversity and functioning of **benthos**	Knowledge	Detailed	Detailed	Detailed	Detailed	Detailed	Detailed
	Severity	Expected C/ May C	Expected C/ May C	Expected C/ May C	Unlikely C	Unlikely C- Expected C	Unlikely C
	Data	SS	SS	SS	Insufficient	SS	Insufficient
	Scale	Finest, rectangle, area, RAC	Rectangle, area, RAC	Rectangle, area, RAC	Rectangle, area, RAC	Rectangle, area, RAC	Rectangle, area, RAC
Biodiversity, size spectra and functioning of the **fish community**	Knowledge	Detailed					
	Severity	Expected C/ May C					
	Data	SS	SS	SS	Insufficient	SS	Insufficient
	Scale	Rectangle, area, RAC					
Abundance of **commercial fish and shellfish**	Knowledge	Detailed					
	Severity	Expected C/ May C					
	Data	SS					
	Scale	Rectangle, area, RAC					
Bycatch and abundance of **marine mammals**	Knowledge	Detailed					
	Severity	Unlikely C	Unlikely C	Unlikely C	Unlikely C/ May C	Unlikely C- Expected C	May C
	Data	SS	SS	SS	Insufficient	Insufficient	Insufficient
	Scale	Rectangle, area, RAC					
Bycatch and abundance of **seabirds**	Knowledge	Sufficient					
	Severity	Unlikely C	Unlikely C	Unlikely C	Unlikely C/ Expected C	Unlikely C/ Expected C	Unlikely C
	Data	SS					
	Scale	Rectangle, area, RAC					

7.6.2 Baltic Sea RAC

Table 7.6.2.1 shows the outcome of WGECO analysis for the Baltic Sea, the capacity of ICES to advise on each interaction and whether the knowledge base to give this advice is adequate in relation to (1) understanding of the fishing effect (2) availability of data to conduct relevant analysis.

Table 7.6.2.1. The effects of fisheries in Baltic Sea RAC métiers on ecosystem components and the preparedness of ICES to advise on those effects based on criteria for knowledge (detailed, sufficient, insufficient), severity (expected to compromise CFP objectives, may compromise CFP objectives, unlikely to compromise CFP objectives), data (sufficient all areas, sufficient some areas, insufficient) and scales at which advice on the impact could be given (finest, ICES rectangle, ICES area, RAC). Further details of criteria in Table 7.5.1.

Effect on component	**Métier:** Issue	**Otter trawl**	**Gill net**	**Fixed gear**	**Small mesh industrial fisheries**	**Pelagic**	**Drift net**
Impact on sensitive **habitats**	Knowledge	Insufficient	Insufficient	Sufficient	Sufficient	Sufficient	Insufficient
	Severity	May C	May C	Unlikely C	Unlikely C	Unlikely C	May C
	Data	Insufficient	SS	SS	SS	SS	SS
	Scale	RAC					
Nutrients	The effects of fishing on nutrient flows and bio-geochemical rates are related to the physical disturbance by gears in contact with the seafloor. The resuspension of sediments that occus during the trawling process may be associated with the release of nutrients (and contaminants) that have previously been stabilised in the sediments. Most of the available literature on nutrient flux is not from the Baltic Sea. Most of these effects will not however be specific to the North Sea. The variance in effects between métiers is related directly to the scale and frequency with which the gear disturbs the seabed and the nature of the seabed over which the gear is being towed. Gears that penetrate deeper into softer sediments will change nutrient flux much more than those that are towed over harder sediments. Gears that do not touch the seabed will be unlikely to affect nutrient flux. Advice can be provided at the Rectangle, area and RAC levels.						
Plankton	To the best of our knowledge there are no significant direct effects of fishing on plankton (phytoplankton or zooplankton). There is some evidence that change in the population size and distribution of sprat and herring that feed on plankton is a significant driver in the structuring of the Baltic Sea zooplankton community in some seasons. Changes in the abundance of clupeids, caused by the direct and indirect effects of fishing, will alter the total amount of *Pseudocalanus elongatus*, a key plankton species in the Baltic. The availability of this species may influence Baltic cod productivity at larval and postlarval stages (EU STORE project). A large stock of clupeids may exert top-down control on *Pseudocalanus elongatus* and heavy fishing could lead to increase in the stock of *Pseudocalanus elongatus*. In many regions the seasonal input of meroplanktonic larvae comprise a major part of the zooplankton and influence system dynamics through their consumption of phytoplankton and microzooplankton. Similarly, there are certainly occasions when large, gelatinous, plankton are caught in or macerated by passage through nets in late summer. We are not aware of any studies that allow us to comment on the ecological consequences of this mortality. Advice can be provided at the Rectangle, Sub- Divisions and RAC.						
Biodiversity and functioning of **benthos**	Knowledge	Insufficient	Insufficient	Sufficient	Sufficient	Sufficient	Insufficient
	Severity	May C	May C	Unlikely C	Unlikely C	Unlikely C	May C
	Data	Insufficient	SS	SS	SS	SS	SS
	Scale	RAC					
Biodiversity, size spectra and functioning of the **fish community**	Knowledge	Detailed					
	Severity	Expected C	Expected C	Expected C/May C	Expected C/ May C	Expected C/ May C	Expected C/ May C
	Data	SS					
	Scale	Rectangle					
Abundance of **commercial fish and shellfish**	Knowledge	Detailed					
	Severity	Expected C	Expected C	Expected C/May C	Expected C/ May C	Expected C/ May C	Expected C/ May C
	Data	SS					
	Scale	Rectangle					
Bycatch and abundance of **marine mammals**	Knowledge	Sufficient	Insufficient	Insufficient	Sufficient	Sufficient	Insufficient
	Severity	Unlikely C	Expected C	May C	Unlikely C	Unlikely C	May C
	Data	SS	Insufficient	Insufficient	SS	SS	Insufficient

	Scale	RAC					
Bycatch and abundance of **seabirds**	Knowledge	Sufficient	Insufficient	Insufficient	Sufficient	Sufficient	Insufficient
	Severity	Unlikely C	Expected C	May C	Unlikely C	Unlikely C	May C
	Data	SS	Insufficient	Insufficient	SS	SS	Insufficient
	Scale	Rectangle					

7.6.3 Other RACs

For the other RACs, the information available to WGECO at the meeting was less comprehensive. For the North-western waters RAC (Table 7.6.3.1), South-western waters RAC (Table 7.6.3.2), Pelagic RAC (Table 7.6.3.3) and High seas/long-distance fleet RAC (Table 7.6.3.4), we provide examples of where we can provide advice on specific known ecosystem effects of fishing. ICES can also provide more generic advice on e.g. trawl effects on sensitive habitats. In some cases, such advice may be able to be made more geographically specific, such as when distribution information exists on the occurrence of such sensitive habitats. WGECO did not have the expertise to advise on the Mediterranean RAC.

Table 7.6.3.1. Examples of the effects of some métiers used in fisheries in North-western waters on ecosystem components and the preparedness of ICES to advise on those effects based on criteria for knowledge (detailed, sufficient, insufficient), severity (expected to compromise CFP objectives, may compromise CFP objectives, unlikely to compromise CFP objectives), data (sufficient all areas, sufficient some areas, insufficient) and scales at which advice on the impact could be given (finest, ICES rectangle, ICES area, RAC). Further details of criteria in Table 7.5.1.

Métier	Demersal trawl on Rockall	Gill net fishery on Rockall	Scallop dredge	Beam trawl Irish Sea	Nephrops trawl	Scallop dredge	All targeting fish
Effect	Impact on cold water corals	Impact on cold water corals	Destruction/ change of mearl beds	Impact on benthic species	Changes in benthic community	Changes in benthic community	Change in size/age structure, genetics and size spectra in the fish community
Knowledge	Detailed	Detailed	Detailed	Detailed	Detailed	Sufficient	Detailed
Severity	Expected C	May C	Expected C	May C	Unlikely C	May C	Expected C
Data	SS	Insufficient	SS	SS	SS	SS	SA
Scale	Finest, rectangle	Finest, rectangle	Finest, rectangle, area, RAC	Rectangle, area	Rectangle, area	Rectangle, area, RAC	Area, RAC
Métier	Pelagic trawl fleet	Bass pair trawl fleet	Bottom set gillnet in SW Approaches	Bottom set gillnet in SW Approaches	Longlines		
Effect	Reduction in blue whiting as a food for other marine organisms	Bycatch of common dolphin	Bycatch of harbour porpoise	Bycatch of common dolphin	Bycatch of fulmars, shearwaters		
Knowledge	Insufficient	Detailed	Detailed	Detailed	Insufficient		
Severity	May C	May C	Expected C	Expected C/ May C	Unlikely C		
Data	Insufficient	SA	SS	Insufficient	Insufficient		
Scale	Area, RAC	Rectangle, area, RAC	Rectangle, area, RAC	Rectangle, area, RAC	Rectangle, area, RAC		

Table 7.6.3.2 Examples of the effects of some métiers used in fisheries in south-western waters on ecosystem components and the preparedness of ICES to advise on those effects based on criteria for knowledge (detailed, sufficient, insufficient), severity (expected to compromise CFP objectives, may compromise CFP objectives, unlikely to compromise CFP objectives), data (sufficient all areas, sufficient some areas, insufficient) and scales at which advice on the impact could be given (finest, ICES rectangle, ICES area, RAC). Further details of criteria in Table 7.5.1.

Métier	Demersal trawl	Bottom gillnets	Demersal trawl	Bottom gillnets	Demersal trawl	Pelagic trawls, longlines, gillnets, seine and fixed gear	Demersal trawl
Effect	Ploughed sediments, turned over rocks, loss/damage to structural sessile species	Damage of structural sessile species	Decrease of richness, diversity and biomass of benthos	Impact on sessile vertical developed species	Decrease of richness, diversity and biomass of fish community	Change in community structure	Decreasing abundance of target species, decrease mean trophic level
Knowledge	Detailed	Insufficient	Detailed	Insufficient	Detailed	Insufficient	Detailed
Severity	Expected C	May C	Expected C	May C	Expected C	May C	Expected C
Data	SA	Insufficient	SA	Insufficient	SA	Insufficient	SA
Scale	Area	Area	Area	Area	Area	Area	Area
Métier	Pelagic trawls, longlines, gillnets, seine and fixed gear	All gears	Demersal trawl	Longline, pelagic gillnets, inshore fixed nets			
Effect	Decreasing abundance of target species	Bycatch of dolphins and reptiles	Bycatch of seabirds	Bycatch of seabirds			
Knowledge	Insufficient	Insufficient	Insufficient	Insufficient			
Severity	May C	May C	Unknown	May C			
Data	Insufficient	Insufficient	Insufficient	Insufficient			
Scale	Area	Area	Area	Area			

Table 7.6.3.3 Examples of the effects of some métiers used in pelagic fisheries on ecosystem components and the preparedness of ICES to advise on those effects based on criteria for knowledge (detailed, sufficient, insufficient), severity (expected to compromise CFP objectives, may compromise CFP objectives, unlikely to compromise CFP objectives), data (sufficient all areas, sufficient some areas, insufficient) and scales at which advice on the impact could be given (finest, ICES rectangle, ICES area, RAC). Further details of criteria in Table 7.5.1.

Métier	All pelagic gear	Pelagic trawl and purse seines	Pelagic trawl and purse seine	Pelagic trawl and purse seine	Pelagic trawl	Pelagic trawl and purse seine	
Effect	Impact on habitats	Herring may exert top-down control on *Calanus finmarchicus*, herring fishing could affect plankton	Any effect on benthos	Changes in fish community	Bycatch of salmon	Bycatch of elasmobranch s.	Bycatch of marine mammals
Knowledge	Sufficient	Sufficient	Sufficient	Sufficient	Sufficient	Insufficient	Insufficient
Severity	Unlikely C	Unlikely C	Unlikely C	Unlikely C	Unlikely C	Unlikely C	May C

Data	Insufficient	Insufficient	Insufficie nt	Insufficient	SS	Insufficient	Insufficient
Scale	Rectangle, area, RAC	Area, RAC	Rectangle , area, RAC	Area, RAC	Area, RAC	Area, RAC	Area, RAC

Table 7.6.3.4 Examples of the effects of some métiers used in fisheries in the high seas/long distance RAC on ecosystem components and the preparedness of ICES to advise on those effects based on criteria for knowledge (detailed, sufficient, insufficient), severity (expected to compromise CFP objectives, may compromise CFP objectives, unlikely to compromise CFP objectives), data (sufficient all areas, sufficient some areas, insufficient) and scales at which advice on the impact could be given (finest, ICES rectangle, ICES area, RAC). Further details of criteria in Table 7.5.1.

Métier	Demersal trawl on Hatton Bank and seamounts	Longlining for Patagonian toothfish and Antarctic fisheries	Demersal trawl	Long-line for large pelagics in Atlantic and Indian Oceans	Pelagic trawl for tuna in Atlantic and Indian Oceans	Longlining in southern oceans	Demersal trawl in southern oceans
Effect	Destruction of cold water corals	Bycatches	Overfishing due to inadequate regulation/ control	Bycatch of marine turtles	Bycatch of dolphins	Bycatches of albatrosses and petrels	Trawl wire impact on albatrosses and other seabirds
Knowledge	Insufficient	Sufficient	Insufficient	Sufficient	Insufficient	Detailed	Sufficient
Severity	Expected C/ May C	May C	Expected C	Expected C	Unknown	Expected C	Expected C
Data	SS-insufficient	SA	Insufficient	SS/ insufficient	Insufficient	SA	SA
Scale	Rectangle (equivalent)	Rectangle (equivalent)	Area (equivalent)	RAC (equivalent)	RAC (equivalent)	Rectangle (equivalent) and larger	Rectangle (equivalent) and larger

7.7 Preparedness of ICES to advise on the ecosystem effects of fishing

Inevitably, this review has not been comprehensive, but important patterns have emerged that provide insight into the capacity of ICES to advise on the ecosystem effects of the fisheries relevant to the RACs.

The first deficiency to note is the lack of knowledge on precise ecosystem effects of many fisheries and métiers. Some of this deficiency may be due to the lack of knowledge within the working group (the most obvious example being our lack of knowledge of Mediterranean issues); in most cases a literature and web search was conducted for relevant information, but it is still likely that some studies will have been missed. However, it more likely that there are insufficient studies to characterise precise ecosystem effects by most métiers. It is notable that the majority of relevant studies in the ICES area are from the North, Baltic and Irish Seas, with relatively few in the south-western waters RAC area. To an extent, studies can be generalised across the RACs – thus trawl impacts on fragile habitats (such as *Lophelia* coral reefs) are likely to be the same regardless of the area within which they occur. ICES is thus probably in a reasonable position to provide generic advice on most ecosystem topics within the ICES area but may need to recruit expertise to ensure local or area based knowledge is incorporated in advice. Outside the ICES area (e.g. Mediterranean, southern oceans), ICES may be able to help other more relevant organisations.

In many cases, the degree and nature of the ecosystem effect will depend to a large extent on the nature and scale of the fishing activity occurring in an area. The principal deficiency we have identified relates to the availability of spatial and temporal information on the distribution of fishing effort by each of the métiers within RACs. The risk posed by this gap is that relatively good ICES knowledge on the biological effects of the fishing impact and the potential severity of these impacts cannot be used to support advice in the absence of knowledge on the spatial and temporal distribution of fishing effort or mortality by métier. For many fishing impacts it should be noted that such effort data need to be available at relatively fine resolution such as provided by VMS (e.g. impacts on small areas of habitat).

Even if comprehensive VMS data were available, in terms of ICES preparedness to provide advice to the RACs on the impacts of fishing on the ecosystem, it is notable that there are very few expert groups in ICES that deal with spatial issues and collate the geographically referenced data needed to support spatial management. A recent example where ICES has successfully used spatial information has been the work of WG deep water ecology (WGDEC) and its collations in relation to fisheries effects on Rockall and parts of the high seas. The capacity within ICES to deal with geographically referenced data needs to be improved significantly as this ability will become increasingly important as spatial management is instituted and advice is needed on the effects of closed areas and other geographically-related fisheries management measures. It is important to understand the limitations, pitfalls and dangers of misinterpretations with spatial data. Advice may also be requested by customers interested in other aspects of marine management (e.g. WKFMMPA).

As noted above, geographically referenced data has not been used widely in ICES. The lack of sufficient expertise in the area of data handling extends also to the more theoretical aspects of such data. Geographic specificity is already included to an extent in many pieces of ICES advice; the availability of better geographic data may change the nature and type of this advice. ICES needs a mechanism to develop the theoretical basis of use of geographically-specific data in order to ensure future compatibility and coherence of advice.

7.8 Métiers as units for assessing impacts and the provision of advice

Métier is a relatively natural dimension within which to provide ecosystem advice as studies have often been conducted by métier (defined by gear, target species and location). ICES is moving towards providing advice on the basis of effort by métier. ICES has long based advice on stocks of fish, delineated by ICES Area or Areas. If advice is provided on fish stocks to RACs, presumably this will be on the basis of ICES Area within each RAC area. As can be seen above, WGECO considers that ICES is also in a position to provide ecosystem effect advice by métier and by geographic area. The level of detail available for each of these dimensions varies as indicated in the tables in this Section.

7.9 Recommendations

1. In future discussions with RACs, ICES to draw attention to its expertise in effects of fisheries on ecosystems. The strongest cases can be made in the North Sea and the Baltic.

2. Relevant ICES WG chairs should be aware of the possibilities that questions on ecosystem effects of fisheries may come to ICES from the RACs. These may come from any of the RACs and WG chairs should try to ensure participation of experts from all relevant parts of the ICES area. For example, WGECO may need more expertise from Iberian peninsula and Macronesian islands.

3. ICES to continue dialogue with EC, NEAFC and national authorities over gaining access to anonymous, fine-scale spatial information on fishing effort, for example detailed VMS data. This should be disaggregated as far as possible within each métier.

4. ICES to continue to develop in-house capacity to process and use spatially-referenced data, including habitat information, VMS data and other fishery information.

5. ICES should continue to develop theoretical frameworks for use of spatially-referenced data in ecosystem studies and elsewhere.

7.10 References

ICES 2004. Report of the Working Group on the Ecosystem Effects of Fishing Activities. ICES CM 2004/ACE:03 176pp.

ICES 2005. Report of the Working Group on the Ecosystem Effects of Fishing Activities. ICES CM 2005 ACE/03. 144pp.

ICES 2006. MCAP Chair's report of a RAC- ICES dialogue meeting, 20- 21 February 2006. ICES Headquarters, Copenhagen. 27pp.

Trimmer, M., Petersen, J., Sivyer, D. B., Mills, C., Young, E. and Parker, E. R., 2005. Impact of long-term benthic trawl disturbance on sediment sorting and biogeochemistry in the southern North Sea. Marine Ecology Progress Series, 298, 79–94.

8 The REGNS integrated assessment of the North Sea ecosystem

Term of Reference

f) *Review and report on the results of the North Sea ecosystem (overview) assessment undertaken by REGNS and prepare recommendations for further or modified analyses made where appropriate. The tables of gridded data used for the "overview" assessment should be checked and where necessary new data (parameters) included and/or existing data (parameters) updated if relevant;*

8.1 Introduction

European stakeholders at the Conference on the Development of a European Strategy for the Protection and Conservation of the Marine Environment, (Køge, Denmark, 4-6 December 2002) defined an "ecosystem approach" as *the comprehensive integrated management of human activities based on best available scientific knowledge about the ecosystem and its dynamics, in order to identify and take action on influences which are critical to the health of the marine ecosystems, thereby achieving sustainable use of ecosystem goods and services and maintenance of ecosystem integrity*. The Regional Ecosystem Study Group for the North Sea (REGNS), established in 2003, is part of the ICES response to the Ministerial Declaration from the Fifth International Conference on the Protection of the North Sea (Bergen, Norway, 2002). REGNS prepares plans for how ICES should contribute to the development of an Integrated Ecosystem Assessment for the North Sea, utilizing the ICES expert group network and databases (ICES, 2004). An Integrated Assessment is viewed by ICES as a process of actions which support adaptive management and the ecosystem approach, and as the combined numerical assessment of data and information from various sources (including monitoring and R&D programmes).

REGNS has proposed a two-stage process for the North Sea Ecosystem Assessment. They identified six assessment themes to advance the overall objective: i. fisheries, ii. chemical pollution, iii. habitats and species, iv. nutrients and eutrophication, v. ocean climate and processes, vi. management and policy issues. The results of the thematic assessments will then be integrated into a single assessment. In May, 2005, a REGNS workshop was held focussing on achieving the need for numerical (objective) integration of data sets and information from various monitoring and R&D sources to which ICES has access (ICES, 2005a). The May 2006 meeting of REGNS marks the end of a three-year process which was to identify the "key" issues and path to take towards completing an integrated assessment of the North Sea.

In 2005, WGECO was tasked with reviewing and reporting on the available data contributions made to the REGNS process by other working groups and with describing their value to an integrated assessment. Unfortunately, comprehensive data contributions and associated detailed metadata were not available for WGECO to review, and so the working group focussed on two objectives:

1) To develop a suitable framework for the Integrated Assessment building on past descriptions within REGNS and elsewhere, and
2) To develop an approach for evaluating and rationalising datasets as soon as they become available.

One of the key requirements of an Integrated Assessment is a linkage between a biological change in the ecosystem caused by a manageable activity through the pressures that these activities cause. WGECO endorsed the need for two matrices; one which associates individual ecosystem components with specific mechanisms of pressure and another which links those mechanisms to the activities which are responsible for them (ICES, 2005b). Both of these tables were provided to REGNS and can be found in the 2005 WGECO report (ICES, 2005b).

WGECO further described criteria for evaluation of REGNS datasets (ICES, 2005b):

1. Ecological components that have many available parameters should not be over-represented.
2. Spatial resolution should be sufficient to allow cause of change to be inferred. This will be different for different sectors in a thematic approach than at the whole integrated assessment level.
3. Different sectors and components operate at different scales. This is reality and will structure how the analysis of disparate datasets is undertaken.
4. The spatial and temporal scale of patterns derived from sampling should be represented by the scale of the integrated assessment.
5. Because the "best information available" will be of different qualities for different ecosystem components, it is crucial that the uncertainty associated with each status and trend indicator is communicated clearly.
6. Sources of bias and error in data, and the magnitudes of these, should be understood and consistent over space and time.

The 2006 request to WGECO from REGNS called for a review of the existing data and comments on the approach to the "overview" assessment given to the 2005 ICES Annual Science Conference. We reiterate that the framework provided to REGNS in 2005 is critical to the compilation of a meaningful database for the North Sea, and this has not yet been implemented. In responding to the 2006 request we have chosen to highlight the risks to successful achievement of the REGNS goals with the present data set in terms of the above 6 criteria, and to organize the metadata into ecosystem components (Table 8.1.1) for reference to related ecosystem objectives (Section 4) and fishing impacts (Section 3) discussed earlier in this report.

Table 8.1.1. Ecosystem component framework used by WGECO.

Ecosystem Component			
1.	Fish		
2.	Cephalopods		
3.	Benthos		
		i.	Benthos
		ii.	Macrophytes
4.	Plankton		
		i.	Phytoplankton
		ii.	Zooplankton
5.	Seabirds		
6.	Marine mammals		
7.	Marine reptiles		
8.	Habitat		
		i.	Water column and bio-chemical habitat
		ii.	Physical habitat

8.2 Review of the REGNS Dataset

WGECO was provided with two spreadsheets extracted from the REGNS database. One file showed the average values for each ICES statistical square (from all years; 1950-2005). [The North Sea has been divided into about 200 squares roughly 30km by 30km in dimension so any one parameter which had been sampled in a square in any one year was itself averaged.] This data constituted the resource file for spatial average for the parameter for each year. A second file provided by REGNS showed the same parameters in terms of a spatial average from all squares for each year from 1950 to 2005. This produced a single value for each parameter covering the entire North Sea for each year. Collectively these files represent the REGNS database, which form the basis of our report.

This database contained 883 data descriptors or parameters, 702 of which are only recorded in one or two of the years of the 56 year series from 1950 to 2005. The problem with this, as acknowledged by REGNS, is that any form of time-series analysis is by definition impossible for such parameters, and therefore they should be removed for such purposes. These parameters pertain to the North Sea benthos and are recorded in 1985 and/or 1986 only. For a static view of the ecosystem, such as that reported in the North Sea Ecosystem Overview, these data are valuable. For the purposes of our evaluation we reduced the REGNS data set to those data descriptors for which there were 3 or more records in the series. This left 181 data descriptors including both biotic and abiotic data types. Subsequent discussion is focussed on this reduced dataset.

8.2.1 Spatial and temporal coverage of records in the reduced REGNS database

There is a considerable amount of spatial and temporal variation in the reduced dataset. The reduced REGNS database (see Section 8.2 above) contains biotic and abiotic data from 227 ICES rectangles (Table 8.2.1.1) which we partitioned into the number of data descriptors for each of the ecosystem components listed in Table 8.1.1. The ecosystem component “seabirds” showed the best spatial coverage ranging from 179 to 208 ICES rectangles (average 207.4). The lowest spatial coverage was observed for biochemical data of the water column. These data also showed the highest variability (3-213 rectangles, average 79) regarding the spatial coverage of individual parameters.

The average temporal coverage of the ecosystem components ranged from 17.3 for the biochemical data to 44.15 years for fish data. Most ecosystem components showed a high consistency in the temporal coverage of datasets, rendering them useful for examining long-term changes in the data.

Table 8.2.1.1. Breakdown of the reduced REGNS data descriptors (parameters) by ecosystem component. The Section of this report dealing with specific issues regarding the data is given in brackets next to each ecosystem component. The number of parameters, and their average, minimum and maximum spatial and temporal coverage are listed.

Ecosystem component (Section)	No. Variables	Aver. No. ICES Rectangles	Min.	Max.	Aver. No. Yrs	Min.	Max.
Fish (Section 8.2.2.1)	20	177	109	212	44.15	41	48
Cephalopods (Section 8.2.2.2)	1	166	166	166	40	40	40
Benthos (Section 8.2.2.3.1)	1	161	na	na	40	na	na
Macrophytes (Section 8.2.2.3.2)	0	-	-	-	-	-	-
Zooplankton (Section 8.2.2.4.1)	36	176.5	157	179	43.4	6	54
Phytoplankton (Section 8.2.2.4.2)	14	178.4	170	179	40	40	40
Seabirds (Section 8.2.2.5)	56	207.4	179	208	25	25	25
Marine mammals (Section 8.2.2.6)	0	-	-	-	-	-	-
Marine reptiles (Section 8.2.2.7)	0	-	-	-	-	-	-
Water column & bio-chemical habitat (Section 8.2.2.8.1)	53	79	3	213	17.3	1	46
Physical habitat (Section 8.2.2.8.2)	0	-	-	-	-	-	-

8.2.1.1 General problems with the existing database

We interrogated the REGNS web-database, as well as the Excel spreadsheets provided, to ensure that the two were consistent, which they appeared to be. In both, we found “DIV/O” entries which we have been told by REGNS should be blank. We also found some anomalous entries which appear to have arisen through data entry errors, and these should be verified. For example the record for 1962 for the diatom *Proboscia alata* is anomalous at 3690817.35, and again at 17280471.6 for statistical square 50E6, as it is eight orders of magnitude above all other records for this taxon. Similarly, the English landings for cod in 1965 (305453.76) is one order of magnitude smaller than those of adjacent cells. It is important to establish whether this was due to fishery closures, data entry error, or biological change as it will have a major influence on the analyses (see Section 8.3.1). The squid landings value for 1972 is two orders of magnitude greater than surrounding cells and should also be verified. In general, examination of the minimum and maximum values for each parameter should be done prior to publication of the data set to address these questions. Real extremes should be noted in the data descriptions so that data users can be confident of their analyses.

The CPR series also has some records which could cause confusion or introduce errors into analyses. The series is broken into two periods with separate data parameters: 1950-1997, and 1998-2005. Lamellibranch larvae data are recorded from 1950 to 1997. From 1997 to 2005 the data are entered as Lamellibran<u>chia</u> larvae and the abundance estimates are an order of magnitude greater. Treatment of these descriptors as a single continuous variable would introduce to any analyses what we assume are artificial changes in abundance. Similarly, combined *Calanus* stages are assessed as *Calanus* copepodites from 1950 to 1997, and then as *Calanus* I-IV from 1997 to 2003. As for the mollusc larvae the two data series differ greatly in magnitude with an average of 0.61 in the former series and 17.5 in the later. The copepod *Centropages typicus* is divided into two data records with miss-typed descriptors. It appears with the correct spelling in the 1950 to 1997 and then as *Centr<u>oo</u>pages typicus* from 1998 to 2005. This might be a problem in reconciling the data matrices and lead to discarding one or other record unnecessarily on the basis of incomplete temporal coverage. It is possible that these parameters were named differently on purpose in order to distinguish the two series. This is where accompanying information on the metadata is critical as users will not be familiar with the details of data sets outside of their own expertise and rely on those with experience to make sure that the data is reliable.

8.2.2 Coverage of records in the reduced REGNS database by ecosystem component

In order to apply an ecosystem approach to management questions in the North Sea it is important that all aspects of the ecosystem be represented, just as it is important that any potential agents of change are quantified. Section 4 of this report ranks human activities according to potential impact for each ecosystem component. This will serve as a guide to REGNS for prioritizing data collection. Section 4.4.4 comments on the suitability of the REGNS database to be used in integrated ecosystem assessments.

We have partitioned the data descriptors in the <u>reduced</u> REGNS database into ecosystem components to identify gaps in coverage (Table 8.2.1.1). For each we discuss limitations to the coverage and make suggestions for improving the data or acquiring additional relevant data. We also, where possible, discuss the most important drivers for inducing change for each component and suggest ways of incorporating information on the activity into the database. We strongly recommend cross-referencing these components with Section 3 and 4 where more details are provided.

8.2.2.1 Fish

Demersal and pelagic fish were identified by REGNS as important parameters to be included in the database. Fishing is an important driver of change in the marine ecosystem (Dayton *et al*. 1995, Pauly *et al*. 1998). Consequently, the database should include data on fish distribution and abundance as well as on fishing activity.

In general, little in the way of information on fishing activity is present on the REGNS database, e.g. distribution of fishing effort. Data on landings from Scotland and England only, and CPUE for inclusive pelagic and demersal fisheries ("Dem-" and "PelFishNPerHrTow") are the only metrics in the database provided.

The fish landings are separated into different records for England and Scotland reflecting different spatial coverage but covering the same time period. This introduced spatial variability does not appear to be considered in the analyses. The spatial separation and temporal patterns could be biasing the REGNS multivariate analyses. The WG has strong reservations over using landings data as a proxy for fish abundance in an ecosystem context, as landings can reflect socio-economic conditions more than they do environmental conditions.

The number of fish species currently represented in the data set is limited (7) and should be expanded. Flat fish data have not been included. Further, data on sandeels would be especially valuable given that catch statistics indicate that it is the largest fishery in the North Sea (Source- STATLANT Database for the North Sea; FAO, 2005). Sandeels are consumed by numerous species of fish, seabirds and marine mammals, including seals and cetaceans (See Section 3.6.1.2.1). Sandeels are therefore potentially an important component in the North Sea ecosystem (Greenstreet, 1996; Greenstreet *et al*. 1998) and should be included in the assessment database.

Another missing component is the elasmobranchs. At least 20 species of sharks have been recorded in British coastal waters. In addition at least 12 species of skates and rays, one species of stingray and 2 species of electric rays have been recorded. Basking sharks in particular are temperature sensitive and could be good indicators of environmental change. Data on elasmobranchs should be included in the data set.

Information on other species, including non-commercial species, would allow changes in community/assemblage composition of fish communities to be interrogated as an important aspect of understanding ecosystem changes (Parsons and Lear, 2001; Beaugrand, 2004; de Young *et al*., 2004; Mantua, 2004), for example, an increase in abundance of horse mackerel (*Trachurus trachurus* L.) has been linked to an increase in the northerly advection of water along the western edge of the European shelf (Reid et al. 2001) and the appearance of species with different biogeographic affinities allows hypotheses to be made regarding changes in the system (Beare *et al*., 2004). Inclusion of the data descriptor "SouthFishNPerHtow" partly addresses this later concern, as it is a measure of incursions of fish with southern distributions into North Sea waters, but subdivision of this record by species would allow for differential responses to be evaluated.

The data series also contains a number of derived indices from the stock surveys. Changes in the proportion of large fish and thus the average weight and average maximum length of the fish communities have been derived. Average size (length and weight) in the demersal and pelagic communities are adequately represented for the species selected. Spawning stock biomass data and recruitment data are omitted. Recruitment is known to respond to ecological effects and is not autocorrelated as is the case for annual SSB data. Therefore it is a good ecological indicator (Brander, 2005), while the later is more susceptible to fishing pressures. The Canadian Eastern Scotian Shelf Ecosystem Status Report (DFO, 2003) and NOAA's

Arctic Ecosystem Overview (http://www.arctic.noaa.gov/detect/marine-overview.shtml) include further metrics which should be included in the REGNS database (Table 8.2.2.1.1).

It should be noted that REGNS appears to have followed some of these recommendations in the dataset for which they produced their "traffic light" pictogram of standardised anomalies of CPR, oceanographic and fisheries data at the ICES ASC and in their 2005 report. However, those data were not entered into the dataset provided to WGECO nor do they yet appear on the REGNS web-access database. Presumably, these will appear in the near future. They include data on 72 parameters, 54 of which pertain to fish or shellfish. The fish parameters include data on SSB, landings, total biomass, fishing mortality, and recruitment for an enhanced list of 20 species (not all data types for each species). Their analysis did not include data on the average size of the fish which was in the REGNS data set. The different nature of those metrics must be taken into account for each type of analysis. Stock-recruitment residuals could also be considered for use.

Table 8.2.2.1.1. Fish Metrics Used in Other Ecosystem Overviews and Relevant to the North Sea.

Fish-Related Metrics
1. Pelagic:demersal fish ratio (based on numbers)
2. Fish diversity (richness)
3. Length-at-age (various fish)
4. Fish community similarity index
5. Community condition index
6. Proportion of area with fish in good condition
7. Recruitment index or estimate
8. Relative F of groundfish
9. Bottom area trawled
10. Shannon diversity index for fish

8.2.2.2 Cephalopds

Data on squid landings are included in the REGNS database. Breakdown of this data into component species would be valuable as some have strong warm water affinities and their distribution could respond to temperature changes. Further, cephalopods accumulate heavy metals which can result in death, and so they also function as good indicators of pollution impacts, particularly as their short life-spans cause the population to respond quickly.

8.2.2.3 Benthos

8.2.2.3.1 Benthos

The removal of all parameters with less than three data points effectively removed the North Sea benthos data from the database. The only remaining data on benthic species were the Scottish *Nephrops* landings. Inclusion of the *Nephrops* data is particularly useful as *Nephrops* respond to parasitic infection which in turn may be environmentally regulated, and to temperature (Field *et al*., 1992; Fariña *et al*., 1989). Benthic fisheries for *Pandalus borealis*, the brown shrimp *Crangon crangon*, blue mussels and cockles exist and their landings might also be included in the database (the later two were included in the REGNS 2005 report but are not yet in the database). However the use of fisheries landings data is extremely problematic as they reflect changes in socio-economic conditions, for example new markets or reduced fishing opportunities on other species, much more than they index environmental conditions.

Other benthic sampling data series do exist for the North Sea: Dove Benthic time series (Buchanan and Moore, 1986; Clark and Frid, 2001; Frid *et al*., 1996) and Tees Bay (Shillabeer and Tapp, 1990; Tapp *et al*., 1993; Warwick *et al*., 2002). The UK also has a National Marine Monitoring Programme that includes inshore benthic stations that have coverage going back over a decade. Although local in nature they could be used in sub-regional assessments.

In future, with the availability of seafloor habitat maps international programmes should be established to follow the status of the benthos associated with each habitat type and not on the basis of an arbitrary grid as had to be employed when we lacked knowledge of the seafloor.

Although benthic communities are more robust, less variable with external changes than pelagic ones, and with a slower response, this ecosystem component presents a wide range of species with potential to respond to physical, chemical and biological pressures.

From the huge database provided by the North Sea Benthos surveys, several other species or groups could be used as ecosystem components provided that a more extensive series on their abundance can be collected. For example:

1. Carnivorous echinoderms- seastars (*Stichastrella*), ophiuroids
2. Deposit-feeding echinoderms- Urchins (*Echinocardium*, *Echinocyamus*, *Spatangus*, *Brissopsis*) and holothurians
3. Bivalve molluscs- *Abra*, *Ensis*, *Chamelea*, *Nucula*, *Timoclea*
4. Gastropod molluscs- *Lunatia*, *Colus*, *Epitonium*, *Turritella*
5. Small crustaceans: amphipods, isopods, etc
6. Crabs: *Liocarcinus*, *Corystes*
7. Other crustaceans- *Callyanassa*, *Upogebia*
8. Deposit-feeding worms- Terebellids, *Magelona*, Spionidae, Capitellidae
9. Filter-feeder worms- Sabellidae, Serpulidae
10. Carnivorous worms- *Harmothoe*, *Glycera*, *Nephthys*, *Lumbrineris*, *Eteone*, *Anaitides*
11. Sipunculans
12. Ascidiaceans- *Molgula*
13. Anthozoans

These groups have a differential response to pressure, for example:

1. Substrate loss: affect to a higher degree endobenthic groups (7, 8, 11, part of 2, 10)
2. Smothering: to all groups, especially to sessile groups
3. Change in suspended sediment: to all groups, especially filter-feeders by clogging of filtration systems
4. Change in water flow rate- especially to filter-feeders (3,9,12) but also deposit-feeders (2,8,11)
5. Change in temperature: all groups varying degrees
6. Change in turbidity: all groups to varying degrees, but especially filter-feeders
7. Change in light regime: all groups varying degrees
8. Visual presence: especially to preferred prey (5,6,7,10)
9. Abrasion/ physical disturbance: all groups, especially epibenthic ones
10. Geographic displacement: all groups
11. Synthetic compound contamination: all groups
12. Heavy metal contamination: all groups
13. Hydrocarbon contamination: all groups
14. Radionuclide contamination: all groups
15. Changes in nutrient levels: all groups, especially filter and deposit-feeders
16. Changes in salinity: all groups
17. Changes in oxygenation: all groups
18. Introduction of microbial pathogens/ parasites: no data
19. Introduction of non-native species & GMOs: all groups
20. Selective extraction of species: all groups through cascade effects

8.2.2.3.2 Macrophytes

Data on macrophytes are not included from the REGNS data set. Macrophytes include both macroalgae and seagrasses, both of which play important roles in the marine environment. Macroalgal communities can be highly productive and also provide habitat complexity for diverse organisms. Seagrasses share these attributes and also increase sedimentation and prevent erosion through their rhizome systems, thus providing a stabilizing influence on coastal habitats. Macrophytes are restricted to coastal water and generally to depths less than 200m. Consequently they may be difficult to incorporate into the REGNS database. Indicators such as the spatial extent of seagrass beds could be introduced as a useful metric.

8.2.2.4 Plankton

8.2.2.4.1 Zooplankton

The large-scale zooplankton sampling programme, the Continuous Plankton Recorder (CPR) project, provides REGNS with a long, useful data series with good coverage over the North Sea area (Warner and Hays, 1994). The CPR is operated by a specially designed net which is towed behind cargo vessels and ferries on regular routes in the North Sea. Other spatially-limited data exist, such as the fixed station sampling programmes of the Helgoland Laboratory in the southern North Sea and the Dove Marine Laboratory (University of Newcastle) in the north-western North Sea (see Clark and Frid, 2001). The REGNS database includes 36 data descriptors for zooplankton.

Beaugrand *et al.* (2003) analysed the 28 most abundant zooplankton species in the North Sea and found a pronounced change after the beginning of the 1980s. He comments on the relative potential of different indicators: “The categories ‘total copepods’ and calanoid copepod biomass are not good indicators of change. Their interpretation is not straightforward, as they can encompass a large number of species. Such indices may not detect changes in the community structure in some circumstances. For example, the total abundance of a taxonomic group can remain stable while the species composition changes. This situation occurred for *C. finmarchicus*, which has strongly decreased in the North Sea while its congener *C. helgolandicus* has increased (Beaugrand, 2003; Reid *et al*., 2003). Thus, these types of indicators may be less sensitive than key species indicators.” Beaugrand (2004).

WGECO concurs with this advice and suggest that REGNS use data on species rather than genera or higher-order taxa as much as possible. This will reduce redundant information and balance the number of metrics related to other ecological components. Those key species may also be suitable as descriptive state indicators in the North Sea (Section 4.3.2).

8.2.2.4.2 Phytoplankton

More than one hundred phytoplankton species or groups have been recognised since the beginning of the CPR survey. However, changes through time have only been examined for a few groups (e.g. Edwards *et al*., 2001). This is primarily because the mesh used in the CPR is coarse and the vast majority of phytoplankton pass through. It is only as the net clogs that individuals become collected. However, the impact of phytoplankton on the mesh stain it, and the colour of the mesh itself has been used (the greenness index) to follow total phytoplankton dynamics. Research has focused mainly on phytoplankton colour, an index thought to reflect phytoplankton biomass (Reid *et al.*, 1998). The REGNS data set includes 14 data entries on phytoplankton, including both diatoms and dinoflagellates. Additional metrics used in other studies (DFO 2003, NOAA) are indicated in Table 8.2.2.4.2.1. The WG also considered that the timing of the spring bloom might be an important indicator to include, as well as information on harmful algal blooms.

Table 8.2.2.4.2.1. Phytoplankton Metrics Used in Other Ecosystem Overviews and Relevant to the North Sea.

Phytoplankton-Related Metrics
1. Diatom abundance
2. Dinoflagellate abundance
3. Diatom:Dinoflagellate ratio
4. Timing of spring bloom
5. Frequency of harmful algal blooms

8.2.2.5 Seabirds

The seabird data is well represented in the REGNS data series with 56 data descriptors. In fact this data component could be considered to be over-represented relative to the other ecosystem components. The seabird species data supplied offer a variety of bird types and feeding habits (e.g. piscivorous, benthic, scavenging, opportunistic). The dataset shows the greatest number of complete cells from 1980 to date – with annual data at the level of ICES rectangles.

There are a few problems with the data that should be addressed prior to analysis. A lack of information is sometimes indicated by a zero rather than a blank cell and this inconsistency should be reconciled. The database includes 12 parameters referred to as "Unidentified" followed by a species name. These data are "best guess" data and are not of the same reliability as the fully-identified species parameters. Similarly, the seabird data includes composite parameters of two or more possible species identifications. This occurred when observers could not distinguish between two similar birds or between many. For example, "lesser-black-backed-/-herring-gulls" means that the observer could not distinguish between the two species but knew that it was one or the other; "unidentified large gull" means just that. Thus a herring gull could appear under the "herring gull" parameter (definite identification), under the "unidentified large gulls" parameter or under the "lesser-black-backed /herring gull" parameter. Decisions should be made on whether the less certain parameters are of value. If they are included in analyses they could have a strong influence on the results depending upon their correlation with other variables. The actual species composition of these general parameters is likely to vary from year to year and location to location.

8.2.2.6 Marine mammals

Marine mammals are missing from the REGNS database and these are known to be important drivers of ecosystem structure. Marine mammals are sensitive indicators of changes in ocean environments. Springer (1998) concluded that fluctuations in marine mammal populations in the North Pacific are entirely related to climate variations and change, and MacLeod *et al.* (2005) suggest that recent changes in cetacean occurrence off NW Scotland are due to climate change.

There is little data on the abundance of cetaceans in the North Sea, as only one quantitative survey has been carried out (Hammond *et al*., 2002). Results of a further survey completed during the summer of 2005 will be available in 2006. In contrast, the North Sea haul-out data on grey *Halichoerus grypus* and harbour *Phoca vitulina* seals is more robust (JNCC, 2003). Time-series of abundances are not available for either species for all parts of the North Sea, but data exists for some regions (Duck 2002, Duck and Thompson 2002, ICES 2001, ICES 2005c, http://www.waddensea-secretariat). This data should be included in an integrated assessment as these seals require offshore waters for feeding and no other seals enter the North Sea solely to feed.

Metrics included in other ecosystem overviews (DFO Canada, NOAA) and of potential value to REGNS are listed in Table 8.2.2.6.1.

Table 8.2.2.6.1. Marine Mammal Metrics Used in Other Ecosystem Overviews and Relevant to the North Sea.

Marine Mammal-Related Metrics
1. Grey-seal (or other) abundance
2. Grey-seal (or other) pup abundance
3. PCB in seal blubber

8.2.2.7 Marine reptiles

Two marine reptiles, the loggerhead turtle *Caretta caretta*, and the leatherback turtle *Dermochelys coriacea* are found in the North Sea. The loggerhead turtle is a cold-blooded (ectotherm) species, relying on the external environment to control body temperature. It is found generally in coastal waters. The leatherback turtle is warm-blooded (endothermic), which is unusual for reptiles. It favours open seas and is capable of diving to depths of 1000m or more. Both species are uncommon in the North Sea but they could be good indicators of climate change as they are seasonally more abundant. Identification of important feeding grounds for sea turtles could warrant specific management actions but data on migrations or foraging activities (that might respond to anthropogenic or climatic change) such as exist elsewhere do not exist for the North Sea. Should this change, inclusion of such data would enhance the REGNS database.

8.2.2.8 Habitat

8.2.2.8.1 Water column and biochemical habitat

The REGNS database includes 53 data parameters related to water column and biochemical habitat. Thirty-six of these are data on sediment biochemical properties and were obtained from the ICES Sediment Chemistry Data.

Information in the preceding sections highlights the data needs for ecological data to determine changes in the ecosystem. In assessing regime shifts in the North Sea system there are a number of valuable climatic indices that should be included in order to identify biological responses to environmental change.

The North Atlantic Oscillation (NAO) is an atmospheric mass whose position fluctuates between polar and the subtropical regions (Visbeck *et al*., 2000). The climate of the North Atlantic is dominated by the NAO, whose variable mass and pressure fields also control the climates of the underlying ocean system and surrounding continents on interannual to decadal time scales (Marshall *et al.,* 2001). The NAO accounts for one-third of the total variance in sea-level pressure over the North Sea (Dickson and Turrell, 2000). The North Atlantic Oscillation Index (NAOI) is used to represent the different phases the NAO and is based on the surface pressure differences between the subtropical (Azores) high and the subpolar (Iceland) low, although Lisbon is occasionally used in place of the Azores. The IMR NORWECOM model data matrix (13 sections x 600 months (1955–2004)) provides a time series of southwards (in) fluxes across the northern boundaries to the North Sea shown as averages for the 1st quarter (January-March) over the period. The northern boundaries are made up of the section between Orkney and Shetland and the western and eastern parts of the Feie-Shetland section. These data could be converted into a metric for measuring the NAO as for the Eastern Scotian Shelf Ecosystem overview (DFO, 2003).

The NAO index is not the sole contributor to understanding atmosphere-ocean interactions in the Northern Hemisphere. While the NAO index is often used, another method of monitoring the circulation of the North Atlantic Ocean is the position, measured by the latitude, of the North Wall of the Gulf Stream (GSNW index).

The Gulf Stream Index (GSI) is an indicator of north-south shifts in the latitude of the north wall of the Gulf Stream between 79ºW and 55ºW (Taylor and Stephens, 1980; Taylor, 1995; Taylor, 1996) and has been linked with ecosystem dynamics in the North Sea System (Robinson and Frid; 2002). Similarly, atmosphere ocean systems outside the North Atlantic region may effect the atmospheric/ocean regime in the North Sea. The Southern Oscillation Index / El Niño Southern Oscillation (ENSO) ENSO is associated with climate variability in many regions around the world including the Northern Atlantic (Luksch *et al.*, 2005, Rodríguez-Arias and Rodó, 2004).

Physical oceanographic and atmospheric metrics included in other ecosystem overviews (DFO Canada, NOAA) and of potential value to REGNS are listed in Table 8.2.2.8.1.1.

Table 8.2.2.8.1.1. Physical Oceanographic and Atmospheric Metrics Used in Other Ecosystem Overviews and Relevant to the North Sea.

Metrics	
1.	Position of the Gulf Stream
2.	Stratification anomaly
3.	Sea level anomaly
4.	Volume of CIL source water
5.	Area of bottom < 3 degrees C
6.	Sea Surface Temperature (SST) anomaly (Satellite)
7.	Temperature of the mixed layer
8.	NAO
9.	Storm index
10.	Mixed layer depth (Z)
11.	Sigma-t in mixed layer
12.	Wind stress (total)
13.	Wind stress (x-direction)
14.	Wind stress amplitude
15.	Wind stress (Tau component)
16.	Salinity in mixed layer
17.	Ice coverage

8.2.2.8.2 Physical habitat

As is broadly described in the literature, benthic communities are strongly controlled by substrate type. At present, there is no information on physical habitat in the REGNS data set (see Section 8.2.2.3.1 above). The box-core data from the North Sea Benthos surveys could be used to determine substrate characteristics. For example, a description of habitat characteristics might include a classification of substrate types present in the zone, e.g. surface occupied by ICES rectangle of hard substrates or soft substrate. Hard substrate could be typified by rocks and stones, rocky bottom with low relief, rocky bottoms with low sedimentary coverage, etc. In the same way soft bottoms could be typified using several sedimentary variables:

1. median particle size (Q_{50})
2. sorting coefficient (S_0)
3. weight percentages of gravel and coarse sands (>500 µm)
4. weight percentages of medium, fine and very fine sands (63-500 µm)

5. weight percentages of silt (<63 µm)
6. weight percentage of organic matter

8.3 REGNS analytical approach

There are a plethora of tests available to analyze species and environmental variables, and these could be more fully explored for application to an integrated assessment of the North Sea in future. There is no prescribed approach for analysing these types of data. Multivariate approaches across ecosystem components include rank correlation techniques between species and environmental (or anthropogenic) similarity matrices (e.g. the BIOENV or BVSTEP routines in PRIMER software), and matching data tables using co-inertia analyses (e.g., the Coinertia routine in ADE4 software). For hypothesis testing, multivariate regressions in various forms could be employed. In assessing ecosystem components in a holistic framework it is important to recall that there will be a differential response-time of the various components to activities or changes - certain ecosystem components will have lag times to consider in detecting ecosystem-level effects (Buchanan and Moore, 1986; Collie *et al.* 2004).

In this section we consider the analytical approaches used by REGNS thus far in their 2005 ICES ASC presentation and their 2005 report (ICES, 2005a). We ignore issues of data quality and instead focus on their analytical approach. Other than the analyses commented on in Section 8.3.1 below, WGECO has previously provided comments (ICES, 2005b) on the strengths and weakness of using the “traffic light” approach presented in the REGNS 2005 report (ICES, 2005a).

8.3.1 Multivariate analyses of biotic data over time

Indirect gradient analysis utilizes the species by sample matrix, and relationships between observed gradients and casual factors are interpreted *a posteriori*. By using this approach important gradients can emerge for which there is no external data (e.g. intensity of past disturbance). The alternative, direct gradient analysis, which involves regression, can only say the degree to which the variables reflecting potential forcing factors explain the species variables. Community analysis of the biotic REGNS data was performed by REGNS using an indirect approach, specifically, detrended correspondence analysis (DCA) (Gauch, 1982).

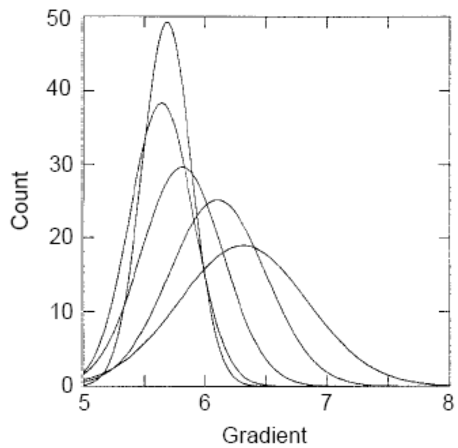

Figure 8.3.1.1. Unimodal species responses to an environmental gradient (taken from Smith 2002).

Another commonly used indirect gradient analysis is non-metric multidimensional scaling (nMDS) which is a species representation approach. DCA is based on an underlying model of species distributions (an unimodal model, Fig. 8.3.1.1) while nMDS is not. Thus, DCA directly relates to theories of community ecology regarding species replacement and disturbance response. However, nMDS is preferable if species composition is determined by factors other than position along a gradient (De'ath, 1999). In the case of the REGNS data set there is no reason to assume that unimodal responses are common in the series. In fact, plotting by REGNS of individual variables in the REGNS data set over time suggests otherwise (Fig. 8.3.1.2).

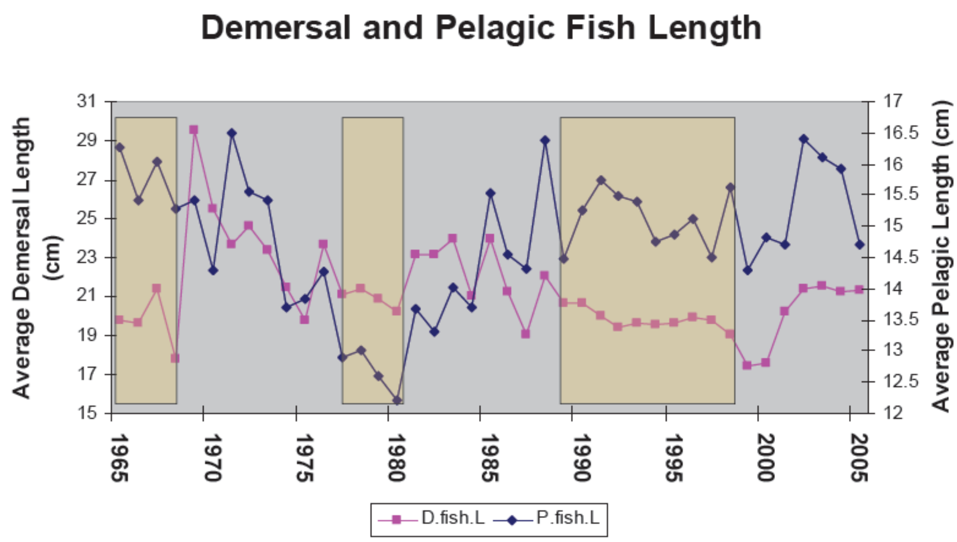

Figure 8.3.1.2. Response of demersal and pelagic fish length variables over time indicating lack of unimodality in the response curves (Kenny ICES ASC presentation).

WGECO extracted the biotic data series from the spreadsheet provided by REGNS and reanalyzed it using nMDS to examine whether the biotic groups created by REGNS using DCA (Fig. 8.3.1.3) were comparable to those produced using a species composition approach. We attempted to analyze the identical database and feel confident that they are highly comparable if not identical. For our analysis, the data matrix was composed of 33 years (1965-1997) and 67 biotic variables for which there were 3 or more data points in the series. Seabirds were not included and the data were square root transformed as was done in the REGNS analyses. To facilitate comparisons, the samples in our nMDS plots were colour-coded using the same colours used in the REGNS DCA.

The two analytical approaches to the common data set gave different results. nMDS showed three discrete groups of samples but they are not the same as those produced by the DCA. Only the 1978-1981 REGNS cluster separates in both analyses, however in the DCA this group is represented by 1978, 1979, 1980 and 1981 samples whereas in the nMDS 1965 is added. The nMDS clusters fall into three consecutive time periods: 1966-1977, 1978-1981 (plus 1965), and 1982-1997 (Fig. 8.3.1.4).

Examination of the taxa which drive the nMDS groupings was done with PRIMER software using the SIMPER routine. That analysis revealed that the fish data are responsible for the observed patterns (Table 8.3.1.1). Specifically, declines in cod and haddock, changes to herring and an increase in mackerel landings characterize the assemblages. Thus the observed patterns are not reflected across the ecosystem components but rather are driven by the fish data. nMDS analyses of individual ecosystem components (fish, plankton and seabird components – not shown) confirmed that only the fish data showed pattern in the series. The inclusion of the 1965 data with the 1978-1981 data in our nMDS may be explained by a data entry error or other issue with the English cod landings (see Section 8.2.1.1 above). This variable accounts for the greatest difference amongst time periods. These results emphasize the importance of reviewing the data prior to analysis and of selecting analyses for which the data are suitable, considering the underlying assumptions of the model utilized.

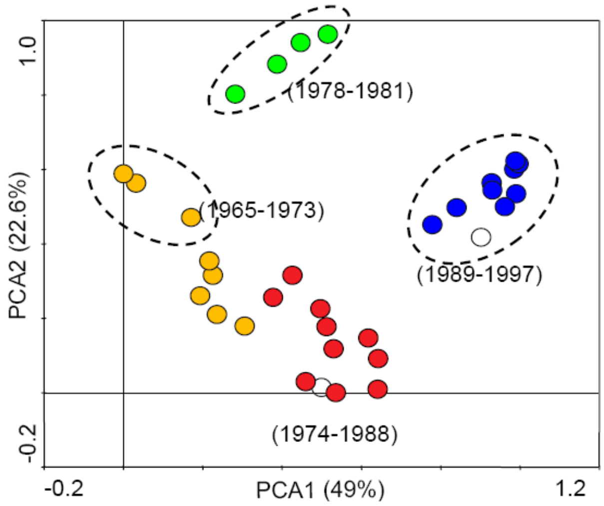

Figure 8.3.1.3. REGNS-produced detrended correspondence analysis (DCA) illustrating colour-coded clusters of the series based on community composition. The three areas of the ordination which represent extreme states (that is they are significantly different from the main trend are highlighted by dotted ellipses. Note the axis labels refer to PCA but they are in fact DCA axes and together they account for over 70% of the variance.

Table 8.3.1.1. Variables contributing to the dissimilarity between nMDS time periods ranked by contribution.

Variable	1966-1977	1965, 1978-1981	1982-1997
Cod- English landings	1731.29	787.32	150.36
Herring- Scottish landings	391.16	21.86	609.23
Haddock- English landings	711.77	383.73	84.72
Mackerel-Scottish landings	46.25	183.79	263.81

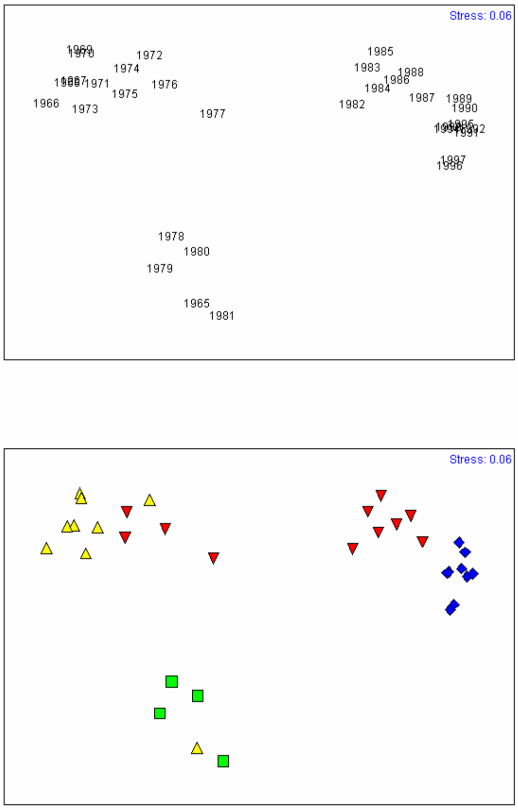

Figure 8.3.1.4. NMDS analysis of the REGNS data set with points labeled by year of collection (A) and colour-coded according to the DCA groups from the REGNS analysis for purposes of comparison (B).

8.4 Options for further integrated analyses

REGNS has proposed a two-stage approach to producing the integrated assessment for the North Sea. This has merit, particularly given the variation in the degree to which data are available for the different proposed themes: i. fisheries, ii. chemical pollution, iii. habitats and species, iv. nutrients and eutrophication, v. ocean climate and processes, vi. management and policy issues. In light of the discussion of ecosystem components in Section 8.2, the ocean climate and processes theme is probably the most advanced in terms of analytical potential. Indeed a number of publications have already appeared examining long-term changes in the plankton in the North Sea (e.g., Beaugrand, 2004).

To progress in other themes, WGECO notes that for many years it has been argued that in monitoring biological entities we have been taking advantage of their biological integration of various natural and anthropogenic influences in the system. For example, there is a large and well known literature on the response of benthic organisms to changes in water-borne contaminants. The benthos will integrate a highly variable spatial and temporal signal of contaminant levels in the environment (e.g., through bioaccumulation). Taking advantage of this and similar ecosystem linkages that integrate diffuse human impacts, there may be numerous opportunities to use selected suites of biological indicators to provide a *de facto* integrated assessment of the state of the environment. What such an approach won't do is identify directly the drivers of change. Section 4 of this report provides an explanation of how to go from the trends in suites of wisely-chosen ecosystem indicators to the types of insights into human (and natural) causes of the trends, that are desired in integrated ecosystem assessments. Such strategies may be of substantial value in making the most use possible of the information that is available to REGNS.

Researchers studying terrestrial systems are using vegetation response models to integrate data on climate change with geo-referenced vegetation data. Such models are used to predict changes under different climate scenarios and can form the basis for hypothesis testing or exploring scenarios about biological, economic, and social impact. ICES may benefit from exploring the use of such models with the REGNS database. Examples include BIOME3 and IBIS, which simulate transient changes in vegetation distribution, biomass, primary productivity and large-scale carbon dynamics, typically over periods of 50-100 years. Details of these models can be found on the German “Register for Ecological Models” website (http://www.wiz.uni-kassel.de/model_db/mdb/biome3.html). WGECO notes, however, that these models focus on predicting how climate affects properties of the plant communities. The ability of these models to simulate scenarios about changes in the animal communities in response to climate change arises from the ability to predict a great deal about terrestrial animal communities from knowledge of their habitats (Tilman, 1999; Morris, 2003). In marine ecosystems aspects of zooplankton communities are predictable from aspects of the phytoplankton (Beaugrand *et al*. 2003). However, in marine communities the concept of “habitat” of higher trophic level species (except for species tied tightly to benthic habitats) is very different from that in terrestrial communities (Bakun, 1998; Rice, 2005), so the ability to use these approaches to modelling responses of the full marine ecosystem to climate change might be more limited. Nonetheless, it is another area worth exploring, particularly in the context of bio-geochemical ecosystem models.

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Annex 1: WGECO terms of reference 2006

2005/2/ACE05 The **Working Group on Ecosystem Effects of Fishing Activities (WGECO)** (Chair: S. Rogers, UK) will meet for 8 days from 5-12th April 2006 at ICES Head-quarters, Copenhagen to:

a) Review and report on the full effects of fishing on the N Sea ecosystem, grouped ac-cording to the suite of ecosystem components identified in previous meetings and where necessary in a regional context, with an emphasis on; i) the direct effects of demersal trawling on benthic species, ii) the ecosystem effects of the small-meshed fisheries targeting fish not for human consumption, iii) the ecological consequences of discarding and iv) the indirect effects of fishery removals on community scale in-dicators identified as promising at past WGECO meetings.

b) Complete the identification and selection of key pressures of human activities on the state of the marine ecosystem begun in 2005, and identify indicators, metrics, data series and reference levels (as appropriate) for these pressures.

c) Examine and take forward recommendations of the Study Group on Management Strategies (SGMAS) (meeting in early 2006) in their review of WGECO suggestions for ways in which ecosystem considerations could be incorporated into fisheries management strategies.

d) Review and report on the work of WGFE 2006 in their further development of the EcoQO on changes in the proportions of large fish and hence the average weight and average maximum length of the fish community, and complete the evaluation of the utility of size based indicators in management frameworks.

e) For each area for which a Regional Advisory Council is established, or is under development, review the preparedness of ICES to advise on the ecosystem effects of the fisheries relevant to the RAC. Where deficiencies are identified, consider the risks posed by the gaps, and suggest feasible steps to redress the gaps in the short or medium term.

f) Review and report on the results of the North Sea ecosystem (overview) assessment undertaken by REGNS and prepare recommendations for further or modified analysis made where appropriate. The tables of gridded data used for the “overview” assessment should be checked and where necessary new data (parameters) included and/or existing data (parameters) updated if relevant.

WGECO will report for the attention of ACE by 15 May 2006.

Supporting information

Priority:	High.
Scientific Justification and relation to Action Plan:	a) The increased focus within the EU and OSPAR on a re-gional approach to the assessment and management of hu-man activities suggests that well targeted reviews of this sort will be a valuable contribution. Specifically, the needs of the N Sea RAC and preparatory work for the OSPAR QSR in 2010, will require a thorough review of the ecosys-tem effects of fishing activity. Colleagues in WGNSSK are now able to deal with this request, made some years ago, and will provide species abundance data (not size composi-tion) for the North Sea Norway pout and sandeel industrial fisheries. This work further develops advice provided by WGECO in 2004 on the ecosystem impacts of industrial fisheries and the bycatch of other trawl fisheries in the N Sea. It also provides an opportunity to build on previous work of WGECO to evaluate community-scale indicators, for which there is an ongoing need to evaluate the indirects of fishing. b) In considering an approach to the integrated assessment of human activities, in 2005 WGECO highlighted the bene-fits of tabulating the pressures caused by human activities, in relation to a full suite of ecosystem components, in order to prioritise the selection of indicators and the imposition of appropriate management action. The ToR further develops this approach by starting the process of identifying indica-tors, metrics, data series and reference levels for human activities so as to populate the table. This work will be an important contribution to the developing European Marine Strategy and will inform the integrated management of the marine environment. c) Output from WGECO 2005 suggested that there was potential for further progress to be made to the integration of ecosystem considerations into fisheries management strategies, during a future joint meeting of WGECO and SGMAS. This ToR will build on the 2006 SGMAS meet-ing and allow us to provide further development of this topic. d) ICES has undertaken detailed evaluation of size-based metrics of fish populations, in support of management proc-esses including the EcoQO framework of OSPAR. This ToR will use the work of WGFE, and that of previous WGs, to finalise the evaluation of size based indicators of fish as performance or surveillance metrics. e) It is increasingly clear that the establishment of RAC is progressing quickly and that they consider themselves to have a central role in the provision of fisheries advice to the EU. For ICES to avoid being marginalised in this process, there is therefore an urgent need for ICES to demonstrate that the science community is fully prepared for the new challenges that this will create. To show that ICES is capa-ble of providing fully integrated ecosystem advice to all RACs, work must begin now. f) This is in direct response to a request from REGNS.
Resource Requirements:	None
Participants:	Approximately 20-25. Wide ranging expertise on fisheries effects and ecosystem components required. Also familiar-ity with EU and OSPAR marine strategies.
Secretariat Facilities:	A large meeting room and secretariat support are required
Financial:	None
Linkages to Advisory Commit-tees:	ACE, ACFM
Linkages to other Committees or Groups:	WGFE, WGDEC, WGMME, WGSE, BEWG
Linkages to other Organisa-tions:	
Cost share	

Annex 2: Recommendations

The following recommendations were made in sections of the report and are for action by ACE & ConC.

(**Section 5**) ICES should identify a couple of suitable test cases for simulation and robustness testing in the face of choices and trade-offs of ecosystem and fisheries objectives, as per the questions in section 5.3.3 of this report and 4.4.2 of SGMAS, and evaluate how well those types of choices fit within the frameworks being developed for evaluation options concerned solely with uses of the target species in the short, medium, and long term. WGECO experts should participate in such evaluations, but they would probably be best led by experts in management strategy evaluations.

(**Section 7**) In future discussions with RACs, ICES to draw attention to its expertise in effects of fisheries on ecosystems. The strongest cases can be made in the North Sea and the Baltic.

Relevant ICES WG chairs should be aware of the possibilities that questions on ecosystem effects of fisheries may come to ICES from the RACs. These may come from any of the RACs and WG chairs should try to ensure participation of experts from all relevant parts of the ICES area. For example, WGECO may need more expertise from Iberian peninsula and Macronesian islands.

ICES to continue dialogue with EC, NEAFC and national authorities over gaining access to anonymous, fine-scale spatial information on fishing effort, for example detailed VMS data. This should be disaggregated as far as possible within each métier.

ICES to continue to develop in-house capacity to process and use spatially-referenced data, including habitat information, VMS data and other fishery information.

ICES should continue to develop theoretical frameworks for use of spatially-referenced data in ecosystem studies and elsewhere.

Future Terms of Reference

The following Terms of Reference have been provided in sections of the report and are reproduced here for further action by ACE and ConC.

ToRs 1 and 2 were provided in section 5 and result from our consideration of the work done in SGMAS.

1. Building on its past work with Ecosystem Objectives, Indicators, and Reference Points, WGECO should select a small number of promising ecosystem indicators for use in management, and test their performance when used with the "classic 3-stage model" for harvest control rules as discussed in WGECO Section 5.3.2.

2. Building on its past work with Ecosystem Objectives, Indicators, and Reference Points, in a research context WGECO should elaborate some candidate decision rules for addressing ecological and fisheries objectives, and test the performance of alternative hierarchical applications of decision rules (as proposed in SGMAS Section 7.2.2.4 and discussed in 5.3.3).

ToRs 3 and 4 were generated in section 6 of this report and relate to the further development of EcoQO for fish size.

3. For the Working Group on Fish Ecology (WGFE)

I. **Review current stock assessments to determine reasonable assumptions for the productivity (recruitment, growth and natural mortality) of fish stocks in their current state for as many stocks as possible in the North Sea.**

II. **Using these estimates of productivity for assessed stocks, combined with reasonable assumptions from life-history theory for non-assessed species, estimate future trajectories of the two fish community size-based indicators (proportion of fish >30cm and mean weight of fish) under a variety of different fishing mortality (*F*) scenarios determined across the fish community to which the indicators is applied (ie the part of the community sampled by the specific research surveys).**

4. For the Working Group on the Ecosystem Effects of Fishing (WGECO)

I. **Review the work of WGFE and consider the results of the analyses carried out.**

II. **Use the results of the analyses to complete the work implicit in section 6.4.2 of WGECO's 2006 report to complete the development of fully quantified and operational EcoQOs for the two size-based indicators. This should include:**

a. **Fully quantified targets for the two indicators**

b. **Estimation of projected time-scales by which these targets might be achieved under a variety of different remedial fisheries management strategies (reductions in "community" *F*).**

ToR 5 was drafted as a result of work done on the identification of key pressures and impacts in section 4, and following previous work on this topic in WGECO. It recognises that there are important tasks still to be completed to advise Client Commissions on how an integrated framework might look, which incorporates the requirements of the EMS to achieve good environmental status (GES), national and international objectives frameworks, and indicator initiatives of ICES, EEA and sectoral management (MSP) in Europe. The international drivers for this work are already well understood and govern the approach taken throughout Europe. Those drivers within OSPAR Annex V (Biodiversity), the European Commission (EMS), and various nature conservation goals (Johanesburg, CBD, Bergen) will be influential, but must operate together and in a coordinated way to deliver sustainability in European seas.

5. Develop an integrated framework for the further provision of ecosystem advice in European Seas drawing on existing experience with implementing the OSPAR EcoQO framework, the implementation of an ecosystem-based approach to fisheries management and proposals for the European Marine Strategy.

<u>Future meeting of WGECO</u>

If requested, WGECO are provisionally willing to meet for 8 days from 11-18th April 2007 at ICES HQ Copenhagen.

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Working Group on Ecosystem Effects of Fishing Activities

ICES Headquarters, 5 – 12 April 2006

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