

8. Annex 3 – The Ecological Disturbance Caused by Fishing

8.1. Introduction to annex 3

Models such as Huston's dynamic equilibrium model consider the effect of disturbance on species diversity. Huston (1994) clearly considered disturbance to be the mortality caused by particular perturbation scenarios to the constituent populations that make up the communities in question, a perception that appears universal among the literature reviewed and cited in Chapter 7. In theoretical ecology terms therefore, disturbance to a community equates to the mortality caused by the "disturbing" events. In attempting to model the effects of fishing activity (the "disturbing" events) on fish and benthic invertebrate communities, it is the mortality caused by the fishing activity that needs to be input as the "explanatory variable", not the measure of fishing activity (eg hours effort per unit area per unit time). The landings data provide an indication of the mortality suffered by the targeted fish species as a result of fishing disturbance.

Ideally, indices of the ecological disturbance caused by fishing should include quantification of total mortality (landings, discards and other) induced by all fishing activities in a given area, over a given time period with weighting for the associated alteration of habitat that occurs (ICES, 2004). Until now, indices of fisheries disturbance have been derived directly from the fishing effort data with no weighting for how factors such as gear type will cause variation in mortality. Fishing effort data should, however, be available for all countries fishing in the North Sea and the number of hours fishing will act as the basic input data for all indices except those that are based only on the landings data. It is considered that two separate but complimentary approaches are required for benthic and demersal fish communities, because their movement rates mean that they are distributed on different scales and because the mortality they sustain due to fisheries is mainly in the catch for fish but mainly on the seafloor for invertebrates. At this time, it is not clear how the consequences of habitat alteration might be incorporated into any indices of fishing disturbance.

In predicting the level of fishing disturbance to demersal fish communities all direct mortality of the species making up that community should be accounted for (including landed fish, discarded fish and fish that die due to damage sustained in contact with the gear). For the main commercial species landings and discards data should be available from all countries fishing in the North Sea. However, discards data are only available for a sample of each fleet and so mortality based on discards would need to be raised to the scale of the fleet. Data on non-target species are less available and in these cases mortality may need to be modelled making basic assumptions, for example, that non-target species have the same catch rates as non-target species with similar characteristics and of the same size class. In this report an index for fisheries disturbance to the North Sea demersal fish community is described based on such an approach (Section 8.7).

In predicting the level of fishing disturbance to benthic invertebrate communities, ideally, as with fish, all direct mortality of the species making up that community should be quantified. However, few data are available regarding the mortality of invertebrates in the catch at the fleet and regional scale. Landings and discards data exist for some commercial species but these make up a very small number of the species that are actually caught and either landed or discarded in North Sea fisheries (see Robinson, 2003 for detail). For non-target species of the benthic community it will be more difficult to predict the mortality sustained in the catch. It is known that a large biomass of non-target

invertebrate species is discarded from North Sea demersal fisheries and using the results of a number of small scale studies of the discards of invertebrates it may eventually be possible to estimate invertebrate mortality from discards.

The majority of the invertebrates that are killed by demersal fishing die as a result of contact with the fishing gear as it passes over the seafloor (towpath mortality) (see Robinson, 2003 for detail). This mortality is not recorded in the catch data because the animals are killed on the seafloor and not caught in the net. In this report a modelling approach is described that will be used to predict the overall mortality of benthic invertebrate communities based on the composition of the species present, the pattern of distribution of fishing effort, the quantity of fishing effort in a described area and the mortality of species per unit of fishing effort (Section 8.6). Benthic invertebrate communities operate on much smaller spatial scales than do fish communities. Thus the microscale distribution of fishing effort can have important consequences on the overall mortality of the community at the scale at which the effort data are available - the ICES rectangle scale (approximately 30x30 nautical miles). The modelling approach adopted takes account of the fact that the micro-scale distribution of fishing effort within ICES statistical rectangles is not even (eg Rijnsdorp et al 1998; Piet et al 2000; in press).

In this chapter therefore, we start by reviewing our current state of knowledge regarding the ecological disturbance caused by fishing activities in the North Sea to the resident fish and benthic invertebrate communities. We then describe the compilation of international landings and effort databases and present the results of basic analyses of these data. Included here is a presentation of the methods employed to model hours-fishing effort data in situations where such information is not routinely reported by the fishermen. Two EC funded projects, the earlier "Biodiversity project (Jennings et al 1999) and the current MAFCONS project, have attempted to compile international effort databases. In both cases, not all countries that prosecute significant fisheries in the North Sea were included in the project consortia, and in both instances it proved impossible to access data from countries outside the project partnerships. In the analysis of the landings data presented here, we assess the relative importance of the fisheries prosecuted by the countries contributing to the MAFCONS landings and effort databases, and those of the countries to whose data we had no access. We conclude that the project partnership included the countries with the most important fisheries in the North Sea, and present a method for raising the MAFCONS data to whole North Sea landings and effort data. Next we present two modelling approaches that utilise the international effort data to estimating the total mortality caused by fishing to the fish and benthic invertebrate communities. From this we derive maps of the ecological disturbance caused by fishing to these components of the marine ecosystem for use in testing Huston's (1994) dynamic equilibrium model (Chapter 2). Finally we also derive maps of spatial variation in the ecological disturbance caused by fishing to the fish community that are based on the international landings data. Again these maps are used to test the dynamic equilibrium model in Chapter 2.

8.2. A review of the disturbance caused by fishing to demersal fish and benthic invertebrate communities

8.2.1. Introduction to review

In the light of the increased urgency to develop ecosystem level understanding of the effects of fishing, it is now highly desirable to be able to predict the community level response of changes in fishing effort that may result from alterations in management. Over the last two decades, there has been a considerable increase in the number of published papers on the effects of fishing at the community level for both fish and benthic invertebrates (For reviews see, Dayton et al., 1995; Jennings & Kaiser, 1998; Hall, 1999; Collie et al., 2000; Kaiser & de Groot, 2000; Johnson, 2002). In most cases however, studies are correlative and descriptive, examining the relationship between a change in the 'level' of fishing effort and a particular community response, such as a change in species diversity, size spectra or species composition of the community. Although these studies provide interesting perspectives into the potential long-term community response there is no means of establishing unequivocally that the disturbance of fishing is the only factor involved.

Both fish and benthic invertebrate communities are structured by a combination of biotic and abiotic factors. These include biotic factors such as competition, predation and larval dispersal and abiotic factors such as climatically driven changes in temperature and productivity (Murawski, 1993; Clark & Frid, 2001; Kröncke & Bergfeld, 2001). In theoretical ecology terms, disturbance is the mortality caused by perturbations to the ecosystem. Thus fisheries disturbance is one of the anthropogenically induced causes of mortality observed in marine systems. In much of the literature describing the structuring of benthic invertebrates however, descriptions are also made of how physical disturbance ranging from the small scale effect of bioturbating animals to the large scale effect of severe storm waves affects the structure of resident communities (Hall, 1994; Auster & Langton, 1999). From this perspective, the mortality experienced by benthic communities, coupled with the change in habitat structure resulting from the passing of the gear is the actual ecological disturbance to the benthic invertebrate community. It is also likely that the change in habitat caused by fishing will have implications for the demersal fish community.

In May 2003, an international workshop was held to discuss how indices could be developed to make predictions about the ecological disturbance of fisheries in North Sea demersal fish and benthic invertebrate communities. At first much confusion was expressed about the difference between the actual ecological disturbance caused by fishing (mortality and habitat change) and the community level changes that are later seen as a consequence of this disturbance (for example a change in the size structure of the community). This confusion reflects the descriptions of fishing disturbance in the scientific literature, where it is generally considered that fishing affects communities both directly and indirectly - one talks of direct and indirect effects. However, when considering disturbance within theoretical modelling constraints, only the direct effects relate to the ecological disturbance caused by fishing. All the indirect effects, the consequences of direct effects, i.e. the changes in competitive relationships caused by the greater mortality suffered by one competitor species compared with the mortality suffered by another, are in effect, the ecological consequences of fishing disturbance. Clearly, to be able to realistically predict the community level response to fisheries disturbance one must first establish the level of mortality experienced by the species making up that community, before inputting this to an overall model of the factors that structure those communities.

The main objective of this review is, therefore, to define the different sources of fisheries mortality in demersal fish and benthic invertebrate communities. Changes in habitat structure that occur following the passage of fishing gear are not implicitly considered at this stage, but the potential to develop disturbance indices that account for the implications of habitat change is discussed. Undoubtedly, levels of fisheries mortality experienced by a community will vary depending on a

number of factors, including the type of fishing gear and power of the vessel being used, the target species of the vessel, and, particularly in the case of benthic invertebrates, the habitat type (Kaiser & Spencer, 1996b; Bergmann et al., 2002; Thrush & Dayton, 2002). For example, it has been found that for each hour of beam trawl effort many more benthic invertebrates and flatfish will be killed than pelagic and demersal roundfish, in comparison with an equivalent hour of otter trawling (Philippart, 1998). As a major second aim, the review will consider the available measures of fishing effort (from the fisheries statistics) in the light of the potential to develop indices of ecological disturbance from them. Until now, indices of the disturbance caused by fishing have been derived directly from the fishing effort data with no weighting added for how factors such as gear type will cause variation in the mortality induced. These indices are usually represented as either a basic scale of the number of hours of fishing in a particular area, or, as the area of the seabed 'swept', or the volume of water trawled in a given time period (see Piet et al in press).

The overall objective of the development of this review is to provide the information necessary to establish what data are needed to develop indices of disturbance to demersal fish and benthic invertebrate communities on a North Sea Scale. This would allow for the prediction of the level of mortality experienced by a particular community given a combination of the number of hours fishing in that area by particular fleets. It is hoped that when the actual indices are developed it will be possible to map the spatial distribution of fishing effort broken down by fleet and gear types, in order to predict the actual ecological disturbance in that area. The available effort statistics will themselves be thoroughly examined to see whether more could be made of the information available, for example by incorporating information on vessel power as well as the hours actually spent fishing by each vessel. The community level response to a particular level of ecological disturbance following fishing activity, will then be based on the interaction of a number of factors, including the level of local productivity, the spatial extent of mortality to individual populations and the history of disturbance regimes in the locality.

8.2.2. Fishing disturbance to demersal fish communities

8.2.2.1. *Mortality of target stocks*

The major targeted demersal fish species in the North Sea are Cod (*Gadus morhua*), Plaice (*Pleuronectes platessa*), Sole (*Solea solea*), Haddock (*Melanogrammus aeglefinus*), Whiting (*Merlangius merlangus*), Anglerfish (*Lophius piscatorius*) and Saithe (*Pollachius virens*). Estimated North Sea landings for these fisheries alone in 1998 were over 336,000 tonnes (Frid et al., 2000), but it has been reported that the landings of demersal fish for human consumption have shown a steady decline over the last 10 years (DEFRA, 2000). Aggregated North Sea demersal landings were shown to be at 38% of the 1970 level in 1996 (Anon, 1998 cited in Frid et al., 2000). The ICES Advisory Committee on Fishery Management (ACFM) produce an annual report that covers all commercially targeted fish and shellfish stocks in the ICES management area. In this report advice is broken down by ICES management regions and by season, if seasonal data are available. Management regions IVa, IVb and IVc cover the North Sea. Data can be extracted on historical trends in landings, spawning stock biomass, recruitment and fishing mortality rates for each individual stock. The reports also provide information on the likely medium term development of the stock using different rates of fisheries mortality and a short term forecast of spawning stock biomass and catch.

Although the ICES assessments will allow the examination of the mortality of target species in the North Sea, data are only generally available at the scale of a sub-region, each of which cover many ICES statistical rectangles. If the aim is to devise indices of ecological disturbance at the smaller ICES rectangle scale it will be important to examine the individual landings for each country fishing in the sub-region as these data are provided at the scale of the rectangle. Using such landings data, it should be possible to determine the mean annual spatial distribution of landings for particular stocks at the ICES rectangle scale across the North Sea. Some preliminary analyses are currently being undertaken at the FRS Marine Laboratory in Scotland to spatially map distribution of mortality of particular stocks based on the catch composition of the trips made by individual Scottish fleets in a given area (Liz Clarke, pers. comm.). Landings data (by year, gear category and ICES rectangle) were provided by most countries participating in the MAFCONS project (England, Scotland, The Netherlands, Germany and Norway) (see Chapter 12 also).

Clearly the data available to calculate fisheries disturbance of target stocks based on landings are of a high resolution. However, these data only provide information on one component of the total fisheries mortality that is experienced by the individual species. A proportion of the target species caught will never be landed and further mortality will be experienced by those individuals that escape the net (escapee mortality as described in Section 8.2.2.3). Large numbers of target species may be caught but subsequently discarded, either because they are below the minimum landing size (MLS) or because the vessel does not have a quota, or is over quota for that species. In some cases, animals that have been caught are later discarded if higher quality specimens are found taking the vessel over quota (a practise known as 'high-grading'). Fishermen have also been observed to discard landable target fish when there is no current market for that species (Cotter, 2003). It will therefore be important to consider fluctuations in marketability of target stocks for those years that disturbance indices will be derived.

Calculating mortality of discarded target stocks could potentially be based on the results of analyses of the data collected by the discarding studies that have been undertaken by fisheries institutes across Europe, supplemented by information gained from individual studies on particular fisheries (e.g. Van Beek et al., 1990). A new EC regulation (more precisely, Article 6(2) of the EC Reg. 1639/2001), states that member states must submit an annual technical report detailing the discard sampling activities of that country (ICES, 2003). A clause in the Data Collection Regulation allows one country to request discarding data from any other member state. In order to derive the discarded numbers it will be necessary to make special data retrievals by individual country, based on the agreement that data are aggregated somewhat for anonymity and linked to the institutes involved (Cotter, 2004, pers. comm.). In most cases, studies of discards on commercial vessels will give total numbers and weights of discarded target species. A number of different 'models' or 'estimators' have been used to then raise the discard mortality from the numbers observed in the observer studies to discard mortality of the total stock in that area (Stratoudakis et al., 1999). It will be important to consider which of these models is most reliable in predicting the discard mortality of targeted demersal fish stocks.

At the same time it should be recognised that these numbers do not represent absolute mortality as it is possible that some animals will survive following discarding. Some field studies have measured discard mortality by holding fish for specific time periods after capture or by using tag and recapture methods, but it has been suggested that the accuracy of such measures is relatively limited (Davis, 2002). There are however data from experimental studies for some target species and it will be important to investigate whether there is enough information to add any level of survivorship to the

discarded numbers when determining the disturbance indices (e.g. Van Beek et al., 1990). Many factors may affect the level of survivorship in discards, including the total time spent on deck, time spent in the codend before hauling and individual species physiological responses to changes in environment (Davis, 2002). Greenstreet & Hall (1996) have even suggested that the apparent long-term increase in the proportion of the flatfish in the demersal fish community of the north-western North Sea may have in part been caused by the preference for roundfish discards over flatfish discards by seabirds scavenging at the stern of fishing vessels (eg. Hudson & Furness 1988). In deriving disturbance indices the resolution of data may only allow that survivorship estimates of discarded species vary by gear, fleet and species.

A number of review papers (e.g. Alverson et al., 1994; Pascoe, 1997; Hall, 1999) have defined the discard problem and suggested possible solutions for the future. Measures suggested to reduce bycatch include the avoidance of areas containing high concentrations of potential bycatch, the modification of fishing gears to reduce capture of bycatch and the modification of gears to allow for escape through grids, panels or increased mesh sizes (e.g. Kennelly & Broadhurst, 1995; Broadhurst, 2000). For some species, numbers of discards are very high and discarding rates appear to be related to the type of gear being used and the habitat type in the fished area (FRS & CEFAS, 2004). Other factors such as changes in legal landing size have also been found to affect the discarding practices for some target species (Stratoudakis et al., 1998). In examining the Dutch Beam trawler fleet, Van Beek et al. (1990) found that from 1976-1990, plaice discards accounted for 49% of the total plaice caught in the North Sea. Total discarded fish mortality has been estimated to be approximately one-sixth to one-quarter of the worldwide fisheries catch (FAO, 1998 cited in Frid et al., 2000; Davis, 2002), with discard mortality representing a large source of uncertainty in estimates of fishing mortality (Alverson et al., 1994; Pascoe, 1997).

Garthe et al. (1996) estimated the total amount of fishery discards in the North Sea to be 262,200 tonnes of roundfish and 299,300 tonnes of flatfish in 1992. These figures were estimated from the landings data for 1992, based on relationships derived from a review of the published data on discard totals for the major fleets operating in the North Sea. This level of discarding amounts to 22% of all fish landed in 1992 and 4% by biomass of the total biomass of fish in the North Sea (landings and biomass estimated by Garthe et al. (1996) from the ICES stock assessment reports). Although there are a number of assumptions made in estimating these levels of discards, Garthe et al. (1996) present some interesting methods for predicting discard levels at the scale of the North Sea and it will be important to consider these when developing the indices of disturbance.

Considerable levels of mortality are experienced by some target fish species that are caught and discarded by fisheries targeting benthic invertebrate species. For example, Frid et al. (2000) suggest that on average, 26% of the catches (by weight) from fisheries in general are discarded, but that in shrimp/prawn fisheries discards are as much as 84% of the catch. However, Garthe et al. (1996) suggest that on a North Sea scale the amounts of discards of fish from invertebrate targeting fleets are low relative to those fleets targeting fish, simply because the fish targeting fleets expend much greater effort overall. However, as fisheries for stocks such as *Nephrops* have developed in recent years, it is important to consider the potential changes in overall discarding of fish that may occur as a result of increasing effort by these fleets. For example, Evans et al. (1994) recorded discards of small Whiting in the Farne Deep *Nephrops* fishery, that were greater than the total catch of the targeted *Nephrops*. On average each vessel was discarding 11,000 undersized Whiting each day! There were also discards of other commercially important fish species and some non-target species and although these were in less significant quantities there are available data for some species.

Revill & Holst (2004) report that the Brown shrimp fishery has long been associated with a high bycatch of juvenile fishes. A European discard study conducted during 1996-1997 estimated that over 900 million juvenile Plaice were discarded in this fishery in a single year (Van Marlen et al., 1998). In a European Commission Report cited in Revill & Holst (2004), it was estimated that such levels of discarding might ultimately result in 7000-19000 tonnes of foregone plaice landings in the North Sea. This equates to 10-25% of the 1998 total allowable catch (TAC) for Plaice in the North Sea and would therefore mean that potentially, in the Brown shrimp fishery alone, an extra mortality of 10-25% of the TAC for plaice has been occurring each year. Clearly such levels of unaccounted mortality must be included in any realistic estimation of the ecological disturbance caused to the demersal fish community. It is also suggested that a number of other commercially important fish species are discarded by the Brown shrimp fishery, albeit to a lesser extent, including Cod and Whiting (Revill & Holst, 2004). It is not however clear whether there are significant discards of any non-target demersal fish such as the common dab. At the community level, it will obviously be important to consider the importance of the discarding levels of all these species in fisheries targeting benthic invertebrates, particularly as high proportions of the discarded mortality are juveniles (Evans et al., 1994; Bergmann et al., 2002; Revill & Holst, 2004).

At the same time, it is important to note that technical measures are currently being introduced, or have already been introduced, to reduce the levels of discarding for some of these fisheries. In some cases these measures have been extremely successful, considerably cutting discarding rates of fish and benthos alike. For example, in trials conducted by Revill & Holst (2004) in the 1999 and 2000 brown shrimp fishing season, the use of a sieve net that directs larger animals such as fish and larger benthic invertebrates out of the trawl net, resulted in a 90% reduction in the retention of unwanted by-catch (both fish and benthic invertebrates), with only an 8% loss of the target species. Of the designs of sieve nets trialled this was the most successful, although all of the designs reduced by-catch by over 56%. In contrast to this however, Bergmann et al. (2002) found that although precautionary measures such as the use of square mesh panels were mandatory at the time of their study in the Clyde Sea Nephrops fleet, undersized commercial fish still accounted for up to 39% of the catch. Undoubtedly it will be important to try to establish both the success rates of the different technical measures, and when the measures have been or are going to be implemented. This will help to predict how the ecological disturbance of such fisheries should change as technical measures are established (See 8.2.4.3). Revill & Holst (2004) observe that sieve nets have been mandatory in the Danish brown shrimp fishery for many years and that all member states of the EU signed up to the implementation of such measures in January 2003.

8.2.2.2. *Mortality of non-target stocks*

A proportion of the catch is made up of non-target bycatch species, some of which is marketable. For the proportion that is marketable, there should be a record of mortality in the landings data, in the same way that there is for the target stocks. A large proportion of the bycatch, however, is not marketable and is discarded at sea. In the North Sea flatfish beam trawl fisheries Garthe & Damm (1997) estimated that 6.6kg of fish were discarded for each kg of sole landed. Frid et al. (2000) suggest these levels of discarding could amount to discarding of 18,000 tonnes of roundfish and 182,000 tonnes of flatfish each year from the beam trawling fleet alone. In calculating mortality of non-target demersal fish, it will be important to try to find records of the mortality sustained by these species in both fish and benthic invertebrate fisheries.

In comparison to the bycatch of non-target benthic invertebrates (described in Section 8.2.3.2), the availability of data on non-target mortality in both the landed catch and discarded fraction is good, particularly in recent years. As described in Section 8.2.2.1, new EC regulations on Data Collection (EC Reg. 1639/2001) have made it mandatory for member states to collect discarding data on both targeted and non-targeted fish stocks. In theory there should be open access to recent data for the levels of discarding of non-target fish stocks from all countries with membership to the EC (J. Cotter, 2004, pers. comm.). However, examination of trends in the discarding of non-target species will be more difficult to access due to a lack of North Sea scale historical data. There is also much less known of the survivorship of discards in non-target species.

8.2.2.3. *Mortality of demersal fish in the tow path of the gear*

Since the mid 1980s research institutes in Scotland and Scandinavia have been conducting experiments that attempt to measure the survival rates of demersal roundfish that escape through the codend mesh during the fishing process (Wileman et al., 1999). Those animals that die following passage through the codend are a source of mortality from the fishing disturbance that is unaccounted for in any of the methods described above. The experiments that have been undertaken into escapee mortality attempt to quantify the proportion of those fish escaping that would die just from the physical injuries and stress induced in the trawling process. This is different to those animals that would suffer natural mortality through predation as a consequence of their weakened state. This latter element of mortality is considered here to be an element of the community level response to the fishing disturbance as described in Section 8.2.5.

Over the last decade considerable improvements have been made in the methodology used in these experiments, with the aim of reducing any elements of the experimental procedure that would lead to increased mortality of the escapees. In a recent EC project investigating the survival rates of roundfish that escape from commercial fishing gear, the most up-to-date methodology were used and it was actually found that their survival rates were conspicuously high in comparison with previous studies (Wileman et al., 1999). It was suggested that the sampling time over which the escapees were collected in the codend cover has a large significant effect upon their survival rates and that in previous experiments this has been a major source of experimentally induced mortality.

In reducing this element of induced mortality, it was found that the estimated mean survival rates of Haddock and Whiting were over 80% in experiments that were conducted with commercial gears. In conventional stock assessments all escapees are currently assumed to survive. Wileman et al. (1999) simulated different escapee mortalities in a stock assessment procedure and found that with an escapee mortality between 10-20% there was a very small impact on the spawning stock biomass per recruit (~1% change). They suggest that introducing these levels of escapee mortality into the stock assessment would not significantly change the result with respect to the perception of the state of the stock in terms of biological reference points. It was however found that escapee mortality in Haddock was dependent on length, and that smaller fish, probably because of their poorer swimming ability, would be less able to avoid injury during their passage through the trawl. It is concluded that management strategies that protect juveniles by improving gear selectivity are soundly based and should be encouraged.

Although these experiments provide important information on the proportion of fish that survive after passing through the trawl gear, the work has only been undertaken on a small number of species from the demersal fish community, in most cases only the commercially important roundfish

species. For these species the work undertaken suggests that this level of mortality is insignificant at the scale of the stock. However, in determining the ecological disturbance of fishing it will be important to consider whether this additional source of mortality is significant at the local community scale. It will also be important to investigate whether there has been any work undertaken on the escapee survival rates of other targeted and non-targeted species.

8.2.3. Fishing disturbance to benthic invertebrate communities

8.2.3.1. *Mortality of target stocks*

The main commercially targeted benthic invertebrate in the North Sea is the Norway lobster (*Nephrops norvegicus*). There are also trawl fisheries for the Brown shrimp (*Crangon crangon*), dredging fisheries for the great scallop (*Pecten maximus*) and the queen scallop (*Aequipecten opercularis*), and potting fisheries for edible crabs (*Cancer pagurus*) and the common lobster (*Homarus gammarus*) (Frid et al., 2000). There are a number of less significant fisheries for bivalves such as *Ensis* and *Spisula* and some emerging fisheries for the crustaceans *Munida* and *Galathea* (I. Tuck and J. Atkinson, pers. comm.). In deriving a fisheries disturbance index of benthic invertebrates it will be important to define whether the index is just for offshore sub-tidal areas or whether it also includes shallow coastal areas. It is likely that indices including shallow coastal areas will need to be parameterised quite differently to more sub-tidally based indices due to the distinction in targeted stocks and gears used. The community level response to the fishing disturbance in these areas is also likely to be different because these shallow areas are subject to much more frequent physical disturbance from currents and wave scour (See Section 5.2).

Data on the landed mortality of targeted invertebrate stocks should be available from assessment and landings records for the more important stocks such as *Nephrops* and scallops. North Sea level data on some of the smaller fisheries may however be difficult to obtain, but in many instances these fisheries only operate in the shallow coastal margins and so there is less need to obtain the data if only parameterising the disturbance indices for offshore sub-tidal areas. Of all the targeted invertebrate species, the most highly resolved information on effort, stock distribution and structure and landings of the fishery are available for *Nephrops*.

Nephrops are exploited throughout their geographical range from Iceland to the Moroccan coast of the Mediterranean. They have been exploited commercially in the North Sea since the mid 1970s and there are important fishing grounds off the northeast coast of England in the Farne Deep and off the northeast coast of Scotland on the Fladden Ground (Evans et al., 1994; Marrs et al., 2000). Annual landings are around 60,000 tonnes and about one third of this is landed into Scotland. TACs for *Nephrops* stocks have been imposed since the 1980s (Marrs et al., 2000) and stock assessments are undertaken annually by the ICES *Nephrops* stock assessment group. Recent EC studies of the North Sea and Clyde Sea *Nephrops* stocks, have however expressed concern at the methods used to calculate the annual stock assessment. This is because the assessments rely on assumptions that are more suitable for finfish stocks, such as homogeneity of the stock, equal capture availability and a finfish behaviour model for redistribution of the stock following capture of part of it (Marrs et al., 2000; Marrs et al., 2002).

Nephrops have been shown to exhibit little homogeneity in their stock size and distribution as their population biology is closely linked to the sediment type and local hydrodynamics (Tuck et al., 1997; Marrs et al., 2000; Bergmann et al., 2002). There are also inherent differences in the

catchability of the different sexes and ages. Berried females and juveniles spend most of the time in the burrow, being much less vulnerable to capture than the more active adult males (I. Tuck & R.J.A. Atkinson, pers. comm.). A new EC study will be trying to resolve the relationship between effort and mortality in the North Sea Fladden Ground Nephrops fishery (Ian Tuck, pers. comm.). It is hoped that the findings of this study will help to improve the stock assessments and thus may provide a more ecologically robust input for target mortality of Nephrops in the invertebrate disturbance index.

Invertebrate fisheries are known to have extremely high discard rates, with the total discards far exceeding the weight of landings (Evans et al., 1994). Much of this discarded material consists of bycatch invertebrates and fish, but there are also substantial quantities of discarded target stock. These are the animals that are either under the minimum landing size (MLS) or those that are discarded because the vessel is over quota and/or better quality specimens above the MLS are found. It is also likely that there will be discards of target invertebrate stocks from vessels operating to target demersal fish. This is further complicated in fisheries that target for both Nephrops and gadoids such as Haddock and Whiting. Under the new EC Data Collection regulation, each member state should be collecting data on discarding rates of all target stocks including invertebrates in all operating fisheries (See Section 8.2.2.1). This may provide invaluable data on the mortality of target stocks not recorded in the landings data.

Of the targeted benthic invertebrate species, there has probably been the most work done on discard rates of *Nephrops norvegicus*. In order to account for the mortality of discarded animals, it is important to know both what proportion of a catch is usually discarded and what proportion of those animals discarded survive. Evans et al. (1994) studied discarding rates of Nephrops in the Farne Deep fishery and found that on average 63.2% by weight (or 85.3% by numbers), of Nephrops caught were discarded. A study by Bergmann et al. (2002) in the Clyde Sea, does however suggest that discarding rates of Nephrops are extremely variable, dependent on the intrinsic effect of sedimentological and hydrological features on the local population dynamics. In the Clyde Sea for example, populations in the northern area have been found to be lower in density than the southern populations, but on average to be of a larger individual size (Tuck et al., 1997). This corresponded with significantly higher discards in abundance of Nephrops in the southern Clyde Sea, due to the higher numbers of small undersized individuals being caught (Bergmann et al., 2002). In a recent EC study, Wileman et al. (1999) found that the mean survival rate for discarded Nephrops was 31%. This is actually higher than that used by the ICES Nephrops stock assessment group who add a discard mortality of 75% to the fisheries mortality used. Wileman et al. (1999) suggest that their study may have underestimated discard mortality, because experimental tows were shorter than commercial tows, thus reducing the time spent on the deck. Another important finding of the study was that the survival rates of discarded Nephrops were significantly lower for females. The implications of this finding should be considered when modelling the population level response to fisheries mortality of Nephrops (See Section 8.2.5.2).

In considering the community level response to the ecological disturbance caused by discarding of target it is important to recognise that discarded specimens are likely to be vulnerable to predation. This will increase the overall mortality experienced by the target stock but will not be included in the actual estimation of fisheries disturbance as predation mortality that follows the discarding event is part of the indirect effects that characterise the community response.

8.2.3.2. *Mortality of non-target stocks*

A proportion of demersally targeted catches is made up of non-target invertebrate bycatch species, some of which is marketable. For the proportion that is marketable, there should be a record of mortality in the landings data, in the same way that there is for the target stocks. A large proportion of the bycatch is not however marketable and is discarded at sea. It has been estimated that between 150 000 to 180 000 tonnes of benthic invertebrates are discarded from North Sea fisheries in a year (Camphuysen et al., 1995; Garthe et al., 1996). This figure includes discards of both target and non-target species. The total amount and catch composition of the discards varies depending on the gears used, what the vessel is targeting and the type of habitat being fished (Bergmann et al., 2002; Lart et al., 2002). In almost all cases, epifauna, followed by shallow burying infauna, are most likely to be captured in the bycatch. Unfortunately, due to the lack of market value, quantification of non-target invertebrate bycatch is rare on commercial vessels and data are only available from research undertaken by a number of institutes over the last 10-15 years (e.g. Craeymeersch, 1994; Fonds, 1994; Ramsay et al., 2001; Bergmann & Moore, 2001; Bergmann et al., 2002). The information that is available from these studies is almost entirely based on either the discarded bycatch from Nephrops trawlers operating in the Clyde Sea (on the West Coast of Scotland), or beam trawlers operating in the Southern North Sea and Irish Sea.

The proportion of the catch made up by non-target benthic invertebrates on Nephrops trawlers appears to be variable and Bergmann et al. (2002) speculate that this is dependent on the characteristic diversity and abundance of individuals in the trawled area, not on the different catchabilities of the gears used in the various studies. In the Clyde Sea study, the difference in catch composition of invertebrates between northern and southern areas was attributed to the differences in heterogeneity of sediments, depths of sites and levels of organic enrichment. This finding suggests that it will be important to have background information on the habitat types, range of depths and levels of organic enrichment in an area for which a disturbance index is being determined.

Quantification of the discards of non-target invertebrates from vessels targeting fish will be even more difficult, as the discards observers on these vessels are not obligated to record any detailed information on this component of the bycatch, often not having the expertise to do so. In most cases there is either no record or only a total weight of the invertebrate bycatch, often referred to as 'trash' (Lart et al., 2002). This may also include non-animal material such as cobbles and shell debris thus making it very difficult to actually determine the level of mortality even at the coarsest taxonomic level. Lart et al. (2002) did however undertake benthic bycatch sampling on a number of vessels operating in the western waters of the English Channel and the southern Irish Sea. Thirty-five hauls from 8 different trips were analysed and the results clearly showed that the type of gear used and the species of fish targeted together explained the separation of hauls into 3 distinct groups of benthic bycatch composition. There were two distinct groups from the vessels using beam trawls, one operating in inshore waters whilst the others operated offshore, and one group of otter trawlers. The beam trawls caught a significantly higher median volume of 'trash' per hour than did the otter trawls, with the inshore small beamers (<9m-beam width) catching slightly more trash per unit volume of fish retained than the offshore beamers. Within these 3 groups it was however possible to detect significant differences of catch composition and the reasons for these differences were more difficult to determine. It was suggested that they could include the effects of the sediment type, time of year and the different specifications of groundgear used within the broader beam trawl and otter trawl categories.

Of the studies that have considered either the discards of invertebrates from invertebrate or fish targeted vessels, there is a common consensus that the volume of this component is often high in comparison to the volume of the marketable catch. It is thus clear from the limited number of studies that have quantified the discards of benthic invertebrates that total abundance and biomass of discarded invertebrates compared to the target stock, are likely to be significant at the scale of the fleet. As with the inclusion of any levels of discarding mortality in a disturbance index however, both the quantities of animals discarded and an understanding of the survivability of the different species following discarding is required. During the work undertaken on the Nephrops fleet in the Clyde Sea, the survivability of a number of the key invertebrate components of the bycatch was studied (Bergmann, 2000; Bergmann & Moore 2001a,b). The survival of the brittlestar *Ophiura ophiura*, the swimming crab *Liocarcinus depurator* and the starfish *Asterias rubens* was significantly reduced, whilst that of a number of other species, including the whelks *Neptunea antiqua* and *Buccinum undatum*, and the hermit crab *Pagurus bernhardus*, was not. The difference in survivability was related to the level of injuries and physiological stress sustained by the different species in the fishing process. Similar findings in a study of the damage to the bycatch of invertebrates in the Manx scallop fishery, indicated that Echinoderms, including starfish and sea urchins, were most vulnerable to high levels of damage, ultimately leading to death (Hill et al., 1996). Hill et al. (1996) also make recommendations on how the mortality in discarded by-catch can be raised to the level of the fleet. This will be helpful in incorporating this element of mortality in the disturbance indices.

8.2.3.3. *Mortality of benthic invertebrates in the tow path of the gear*

A significant fraction of the benthic invertebrates that suffer direct mortality due to fishing are killed as a result of contact with the fishing gear as it passes over the seafloor. This is a much more important source of fisheries mortality to invertebrates than it is to demersal fish due to the largely sessile nature of benthic invertebrates. This ‘unobserved mortality’ is difficult to quantify and it is only in recent years that real progress has been made in bringing together the results of a number of disparate studies (Collie et al., 2000; Johnson, 2002; Kaiser et al 2006).

The only methods that can really be employed to quantify the absolute mortality in the towpath are through counts made by divers, or from remote video or submersibles following the passage of the gear (e.g. Caddy, 1973; Eleftheriou & Robertson, 1992; Hall-Spencer et al., 1999). Even then it is often difficult to establish whether an animal is actually dead or just damaged. Clearly if an animal is badly damaged it is likely that it will be vulnerable to predation or disease as a result of its injuries and thus will face secondary mortality as a consequence of fishing (Hill et al., 1996). However any subsequent predation mortality is an indirect effect of the fisheries disturbance and should not be counted in the quantification of the actual fisheries disturbance index (See Section 8.2.5). Where observed mortality cannot be quantified immediately, studies have calculated the percentage change in abundance, biomass or density of individual populations or communities, either before and after a fishing event or at fished and unfished (control) sites (See Lindeboom & de Groot, 1998 and references in Collie et al., 2000 and Kaiser et al., 2006).

There is an inherent difficulty in interpreting the actual mortality (fishing disturbance) resulting from the fishing event in these studies however, as there is often a time lag between the disturbance and the subsequent quantification of the invertebrate community. This allows for the incorporation of other community structuring factors such as predation, changing resource availability and

immigration of animals into the disturbed area. Thus, the longer there is between the fishing event and the post-fishing sampling event, the greater the likelihood that you are actually measuring the community level response to fishing, rather than the absolute fishing mortality. A number of studies have tried to reduce the effect of this on the interpretation of the actual fishing disturbance. For example, Bergman & van Santbrink (2000) attempted to estimate the annual fishing mortality of megafaunal invertebrate populations in the Dutch sector of the North Sea. To minimise the influence of dispersal on the interpretation of the change in populations following a fishing event, only species that lead a predominantly sedentary lifestyle were included. Also, all sampling of the densities of animals following trawling was undertaken between 24-48 hours after trawling in order to reduce the interference of other biotic and abiotic factors on the estimation of fishing mortality. There was, however, no attempt to try to exclude the effect of predation of damaged animals on the estimation of fishing mortality. It is likely that quantification of the level of mortality of invertebrates in the towpath of the gear that completely excludes any subsequent predation mortality will be difficult to do. Another factor that will make it difficult to gain an accurate estimation of mortality in the towpath of the gear for disturbance indices, is the influence of disturbance history on the level of mortality sustained by populations. It is widely believed that the highest levels of mortality will be sustained in an area that has not been trawled recently. If an area has however been recently trawled, absolute mortality within a population is likely to decrease with each subsequent pass of the gear.

Of the studies that have tried to quantify mortality sustained by invertebrates in this way, it is clear that vulnerability to fatal injury varies dependent on a number of factors. These include life history, ecology and physical characteristics of the biota present (Bergman & van Santbrink, 2000; Piet et al., 2000). There is, however, some disparity between individual studies in the definition of which taxa are particularly vulnerable. This may be because a taxon will be vulnerable in one respect, for example having soft body parts with little armour, but will have this offset by another characteristic such as its' location within the sediment. For example, it is widely believed that thin-shelled molluscs and some echinoderms, such as delicate sea urchins and heart urchins, are at greater risk to serious physical damage than thick-shelled molluscs or robust crustaceans (Rumohr & Krost, 1991; Collie et al., 2000). However, mobility and position within the sediment is equally important in determining their sensitivity. Animals that can quickly retract below the surface, or that live below the penetration depth of the gear will be much less susceptible than epibenthic or near-surface living organisms (Bergman & Hup, 1992; Johnson, 2002). Furthermore, flexibility can also be important in minimising vulnerability to mortal damage, particularly for epifauna (Eno et al., 2001).

There is also evidence that the mortality of benthic fauna in the path of the trawl is strongly size dependent (Engel & Kvitek, 1998; Kaiser et al., 2000; Bergman & van Santbrink, 2000; Duplisea et al., 2002). It is suggested that, within and among species, the mortality rates suffered by the smallest individuals may be lower because they may be pushed aside by the pressure wave in front of the trawl (Gilkinson et al., 1998). This is only relevant, however, to the organisms that are epifaunal in habit or that live close to the surface of the sediment. For animals that are truly infaunal in habit, mortality of smaller individuals may actually be higher if there is a relationship between individual size and depth distribution. For example, Bergman & Hup (1992) found that the fishing mortality of small *Echinocardium cordatum* (Heart urchins) and *Lanice conchilega* (Sand mason worms) was much higher than the mortality in larger individuals. In studying the relationship between depth distribution and size of heart urchins, smaller individuals were found to have a mean depth distribution of 2-4cm, whilst the larger individuals had a mean depth distribution of 10-12cm, below the penetration level of the gear. It is clear that there will be difficulty experienced in trying

to estimate the mortality sustained by each invertebrate population in the towpath of the gear due to the effect of the combination of the various characteristics that influence vulnerability. This is further complicated when the variation in actions of fishing gears and the influence of substrate type are taken into account (See Section 8.2.4.5).

Given the number of variables that appear to affect the population level mortality, the only viable way of determining towpath mortality is to bring together the results of all the disparate studies and to then analyse these to try to pick out consistent patterns. In recent years a number of individuals and projects have undertaken this task. In some cases studies are reviewed and conclusions drawn purely on qualitative evaluations of the combined results (e.g. Watling & Norse, 1998; Johnson, 2002; Thrush & Dayton, 2002). Collie et al. (2000) have however quantitatively analysed the combined findings of these studies using meta-analysis techniques and their work has now been updated and further developed by the EC Cost-Impact project (EC Project: Q5RS-2001-00993) (Kaiser et al., 2006). It is hoped that this work will lead to the potential to predict mortality of invertebrates in the towpath at a population level, given a particular fishing regime, with particular gears, in a specific habitat type. It is important to recognise, however, that only some of the studies included in these meta-analyses actually give absolute fishing mortality values. As described earlier, in many cases there is a delay between the fishing event and the measured mortality in the population, allowing for the incorporation of the community level response (See Section 8.2.5.2). If this significant source of fishing mortality is to be included in the determination of fisheries disturbance to benthic invertebrates it may necessary to make a number of assumptions about the level of mortality actually attributable to the fishing event. It is also very unlikely that it will ever be possible to make these sorts of predictions for all species that make up the diverse benthic communities of the North Sea. However, it is hoped that the science will develop towards the ability to predict mortality for characteristic species and associated functional groups if a particular gear is used in a particular habitat. This will enable a more ecologically meaningful inclusion of towpath mortality in the estimation of fisheries disturbance indices for benthic invertebrate communities.

As described in Section 8.2.2.3 for fish, there is also another element of unaccounted mortality for benthic invertebrates in the towpath of the gear. This is for those animals that actually pass through the fishing gear but then die merely as a result of the injuries they sustain in this process. As with fish, the experimental work that has been undertaken to try to quantify the proportion of escapees that die, has been restricted to commercially important species, in most cases one species, *Nephrops norvegicus*. Wileman et al. (1999) investigated the escapee mortality rate of *Nephrops*, with the assumption that survival rates may be lower than had been found for roundfish fisheries (see Section 8.2.2.3) due to the high quantities of abrasive material usually found in the codend of *Nephrops* trawls. This material includes shells, stones and various crustacea that are mixed in with the target species. Survival rates of *Nephrops* were however found to be comparable with roundfish, with a mean survival rate of 82%. In comparison with the mortality of discarded animals (mean 31% survival rate), the additional mortality associated with escapees from the gear was found to be insignificant from a stock assessment perspective. It is also suggested that the escapee mortality recorded in this experiment may be overestimated as the escapees were held in the codend cover for the duration of the trawl (2 hours). In the same study it had been found that there was a positive relationship between time spent in the cover and increased escapee mortality for roundfish (See Section 8.2.2.3).

In considering the actual ecological disturbance associated with particular gears, it will be important to consider whether this level of additional mortality makes a significant contribution. Certainly in considering the community level response following the fisheries disturbance (Section 8.2.5), it is likely that the escapees will be more vulnerable to predation. In the experiments conducted by Wileman et al. (1999) on Nephrops escapees, it was found that the tail flip mechanism (the escape response exhibited by Nephrops) was reduced by 53% for the first 2 hours following escape from the codend.

8.2.4. Predicting the ecological disturbance of fishing using fishing effort statistics

8.2.4.1. Mapping fishing effort on a North Sea scale

Greenstreet & Rogers (2000) stated that fishing effort has never been evenly distributed across the North Sea. Different gears, directed at different target species, with differing levels of impact on the components of the ecosystem, have been used at varying intensities across the North Sea. In order to develop spatially and temporally resolved indices of the ecological disturbance of fishing on benthic fish and invertebrate communities, at the very least there is a clear need to obtain data for the amount of fishing effort in a given area (eg ICES rectangle or smaller spatial units) at a given time. As described in Sections 8.2.4.4 and 8.2.4.5 below, more resolved information on the types of gear and the power of the vessels used would further improve the potential for developing ecologically meaningful indices. Most of the countries that fish within the North Sea record routine measures of fishing effort at the scale of the ICES rectangle. These data are however variable in the procedures and measures used to record the data and the length of time for which they are available (Greenstreet et al., 1999).

The longest time series available is for the effort of UK vessels landing in Scotland and Greenstreet et al. (1999) analysed trends in both the demersal and pelagic fleets over the period 1960-1994. This work has recently been updated to include the years 1995-1998 (Greenstreet et al., 2006). Jennings et al. (1999b) analysed international trends in demersal trawling over the shorter period 1977-1995. This analysis included effort data from English, German, Norwegian, Scottish and Welsh vessels over the entire period and also Danish and Dutch vessels between 1990-1995. No data were available from the Belgian and French fleets and it was felt that effort had been underestimated, potentially by >50% in the Southern North Sea area, due to the lack of these data. This international database has since been updated to include the year 1998 and the spatial distribution of effort in the North Sea for this period is given in Callaway et al. (2002).

Clearly in trying to develop spatially and temporally resolved indices based on fishing effort statistics it will be necessary to include the data for as many of the countries that fish in the North Sea as is possible. It appears that access to Belgian and French data will continue to be blocked, but if the effort from these fleets is missing, there must at least be an investigation into the proportion of effort missing in given areas. Of the countries that will provide data it is likely that each individual country will have their own system for aggregating the fishing effort per vessel into a number of different gear codes or categories. In compiling an international dataset, Jennings et al. (1999b) found it necessary to combine the gear categories of each country to a common denominator, leaving only two codes, Beam trawlers and Otter trawlers. The potential for resolving gear codes further for particular fleets should be considered in determining the associated ecological disturbance and it may be possible to determine the disturbance in a given area based on the

summation of effort from different fleets operating in that area. This would allow for the availability of different levels of resolution of the effort data.

8.2.4.2. *Improving the accuracy of predicting effort distribution*

In the North Sea the national fishing effort statistics based on logbook data are given at the scale of the ICES rectangle (30 x 30nm). A number of studies in different sea areas have tracked the microscale distribution of fishing effort and it is clear that vessels do not fish at random; in many areas effort is highly aggregated (Churchill, 1989; Pilskaln et al., 1998; Rijnsdorp et al., 1998; Friedlander et al., 1999; Jennings et al., 2000). There are a number of reasons why fishers may operate in a non-random fashion, including the patchy distribution of target stocks and the avoidance of grounds that are either prohibited or dangerous to operate in (e.g. wreck sites, stony grounds, shipping lanes) (Rijnsdorp et al., 1998). In the North Sea, much of the published microscale distribution work is based on the Dutch beam trawl fleet operating in the Southern North Sea (e.g. Rijnsdorp et al., 1998; Piet et al., 2000). Rijnsdorp et al. (1998) estimated that between 1993 and 1996, for the eight most intensively trawled ICES rectangles of the Dutch sector, a mean of 62% of the area was trawled 1-5 times per year, whilst 29% was trawled less than once per year, and 1% was trawled 10-50 times per year. Based on this work, it was also suggested that distribution of effort within the North Sea only becomes random at the scale of 1x 1nm. Thus, at any scale above this, including at the scale of the ICES rectangle, effort is non-random and, as suggested in Rijnsdorp et al. (1998), the highest levels of effort may be aggregated in small areas.

These findings have real implications for the derivation of meaningful ecological indices of fisheries disturbance at the scale of the ICES rectangle. If, for example, a particular level of disturbance (predicted from the mean annual effort in hours fishing of that rectangle) is distributed evenly across the rectangle, areas that may in reality only be subject to very low effort may be overestimated, whilst areas of aggregated high effort may be underestimated. Rijnsdorp et al. (1998) cite a study by Rauck (1985) that predicted the level of beam trawling effort at the scale of the ICES rectangle. This study suggested that every square metre of the seabed was on average trawled 5-7 times per year. However, Rijnsdorp et al.'s (1998) model of effort distribution, based on the microscale data for the beamtrawl fleet, predicted that on average, only 1% of the area within a heavily fished rectangle was trawled more than 5 times a year. Given that it is thought that the percentage mortality sustained by an invertebrate population in the towpath of the gear is likely to depend on the frequency of trawling (See Section 8.2.3.3), estimating a reasonable level of mortality from that source is really dependent on a realistic distribution of effort. A number of studies have now used higher resolution effort distribution data to evaluate the disturbance of fishing in benthic invertebrate populations of the southern North Sea (Bergman & van Santbrink, 2000; Piet et al., 2000). It is clear from the results of these studies that the inclusion of high-resolution effort data significantly effects the estimation of the levels of mortality experienced by populations and communities at the scale of the ICES rectangle.

A number of different sources of data are now becoming available to track the microscale distribution of individual fleets. A proportion of the Dutch fleet has been tracked for over 10 years. Initially 'black boxes' (automated position recording systems) were installed on 10% of the fleet and these gave positions every 6 minutes to an accuracy of approximately 100m (data from 1993-2000). Since 2000 however, the microscale distribution of approximately 30% of the fleet has been available through a private agreement on access to VMS (European Community Satellite Vessel Monitoring System) data (G. Piet, et al in press). Since the 1st January 2000 it has been compulsory

for EC registered fishing vessels over 24m to report their location every 2 hours, using the VMS system. Exceptions include vessels that undertake trips of under 24 hours, or that fish exclusively within territorial waters. Due to problems instigating the system on an international scale, reliable data are only available from July 2000 (Dinmore et al., 2003).

Although VMS data are being recorded by each EC country with a fishing fleet operating in the North Sea, access to the data for scientific research purposes is not always possible. The Dutch data are available for 30% of the fleet and the German data are available for the whole fleet, but access to the data from other countries is more difficult (S. Ehrich, S. Jennings, P. Kunzlik & G. Piet, pers. comm.). It is known that VMS data from both Scottish and English fleets are restricted but it is not known whether there is any access to data from Belgium, Denmark, France or Norway. Another source of information on effort distribution is the overflight data, which is based on the positions of vessels taken by aeroplane observers twice a week (Jennings et al., 2000). This is potentially available for all boats fishing in UK waters and may help to resolve effort distribution where VMS data is not accessible. A number of smaller scale studies of microscale effort distribution also exist for Nephrops targeted fleets in the Clyde Sea and the Fladden Ground of the North Sea (Marrs et al., 2000 & 2002; J. Atkinson & I. Tuck, pers. comm.).

8.2.4.3. *Changes in fishing practices over time*

Technological developments in the fishing fleet of the North Sea have had a profound effect on the types of fishing gear and the power of the vessels used over the past century (Philippart, 1998). Improvements in vessel design and technology have enabled fishing boats to tow larger and heavier gears, to travel faster and to stay at sea for longer. As a result, long-term changes in fishing effort reveal a complex pattern of spatial and temporal interactions (Greenstreet et al., 1999; Jennings et al., 1999b). In order to develop indices of ecological disturbance, it will be important to try to account for any changes in characteristics of the fleet. These may include: changes in efficiency of the gears used; shifts in dominance in the gears used at the scale of the fleet; size and horsepower of vessels making up the fleet (Jennings et al., 1999b). Developments in the fishing power of individual countries are available for some fleets (e.g. Polet et al., 1994; Rijnsdorp et al., 1998; Greenstreet et al., 1999; Rijnsdorp et al 2006).

As an example, the development of the Dutch beam trawl fleet over the last 40 years demonstrates how the interpretation of effort data can be complicated. The Dutch beam trawl for flatfish began just after the Second World War, but effort remained insignificant until the beginning of the 1960s, reaching a peak in the late 1980s (Philippart, 1998; Bergman & Hup, 1992). The maximum number of beam trawlers actually occurred earlier around 1970, but this did not coincide with the peak in effort, as although there were fewer vessels by the late 1980s, the level of effort per individual vessel had increased (Rijnsdorp & van Leeuwen, 1994). Also, it has been reported that both the weight of the gears and the towing speed of beam trawlers were lower in the 1970s, which has implications on the associated mortality of animals both in the gear and in the towpath of the gear (Bridger, 1972; Bergman & Hup, 1992; Jennings et al., 1999b). This point illustrates the importance of considering what the information from the fisheries statistics is actually showing. In developing indices of ecological disturbance based on this information it is likely that the overall level of effort per vessel and the types of gears used, will have more of an effect on the mortality induced than the actual number of vessels at sea. However, a smaller number of vessels may cover a smaller area or be more homogenous in their distribution, thus reducing the spatial scale of the associated ecological disturbance.

It will also be valuable to study changes in spatial distribution of the effort of different fleets over time and to try to interpret why these changes have occurred (e.g. Rijnsdorp et al., 1998; Greenstreet et al., 1999; Jennings et al., 1999b; Greenstreet et al., 2006). If it is possible to associate a change in spatial distribution with the introduction of a new target stock or fishing ground for example, it will be much easier to make predictions of the future spatial distribution of the disturbance associated with individual fleets. However, if fleet distribution is affected by a combination of these factors and others, including target stock size and distribution, and the market value of stocks and price of fuel, the interpretation of how effort may be re-distributed following the change in any of these factors may be difficult. Studies of the spatial distribution of fleets suggest that they are relatively stable over short time periods (e.g. <5 years), but that they may vary quite considerably over longer time periods (e.g. 10-20 years) (Rijnsdorp et al., 1998; Greenstreet et al., 1999; Jennings et al., 1999b; Greenstreet et al., 2004). The frequency of changes in effort distribution may also be variable between fleets, with those that operate more as mixed target fisheries being more variable in distribution over shorter time scales than those that operate for single target stocks.

Finally, the introduction of technical measures within the management of fisheries may also complicate the determination of disturbance indices based on effort statistics. As steps are taken to reduce bycatch, the introduction of alterations to the gear that will help increase the selectivity for landable target stocks are likely to proliferate. Clearly if these measures change the mortality experienced by some elements of the fish and benthic invertebrates communities, it will be important to try to adjust indices of disturbance for those fleets that use them (Revill & Holst, 2004).

8.2.4.4. *Predicting demersal fish mortality from fishing effort statistics*

The determination of the level of fishing disturbance to demersal fish communities in a given area will depend on the ability to accurately predict mortality from the fleets operating in that area. In reviewing the disturbance of demersal fish by fishing, it is clear that there are a number of sources of mortality that must be accounted for. These are the mortality of all fish, both target and non-target that are landed, the mortality of discarded bycatch, also including target and non-target species, and the mortality of fish that escape from the gear but subsequently die (See Section 8.2.2).

For a given number of hours fished (fishing effort), the levels of mortality of the individual species in the landed catch will depend on a number of factors. These may include which stock(s) the fleet is targeting, the market values for each of the marketable species and also potentially fluctuations in stock size and distribution. The selectivity of the different gears used to target particular stocks will also be likely to affect the relative proportions of species in the catch and thus the mortality sustained by individual populations. A study currently being undertaken by FRS Marine Laboratory - Aberdeen, is looking at the characteristic catch composition of vessels targeting specific stocks (Liz Clarke, pers. comm.). This may help to predict the levels of fishing mortality likely to be experienced given the areas that a vessel is fishing in and the stocks it is targeting. Landings data do exist for each of the fleets fishing under EC regulations in the North Sea. It will be important to explore the relationship between the effort of a given fleet and its reported landings in order to try to develop a relationship that can be used to predict this element of fisheries disturbance. Spatial resolution of these relationships will also help to map levels of mortality across the whole area that a given fleet covers. Clearly, in predicting landed mortality based on the historical relationships between effort and landings it is important to realise that official landings data may be inaccurate if

any miss-reporting has been occurring. If possible the prevalence of miss-reporting within individual fleets should be examined.

In order to determine levels of mortality of demersal fish discarded by the fleets fishing in a given area, both the levels of discards per species and the survivorship of those species should be known. For recent years, the numbers of each fish species discarded per trip, should be recorded in the discard monitoring schemes of each country, with a registered fishing fleet in the EC. However, these schemes only cover a sample of all trips carried out by a fleet in a year and so mortality of each species would have to be raised to the scale of the fleet. In order to predict the level of mortality to the demersal fish community sustained in the discards, it will be important to consider the influence of a number of factors on the relative abundance of different species making up the bycatch. For marketable species, changes in quota and market value are likely to affect the level of discarding, whilst the habitat type of the area fished and the targeted stock of a fleet is thought to affect levels of overall discard mortality. Although numbers of discards do not represent absolute mortality of those animals, it appears that there are only a limited number of studies of the survivorship of species following discarding. The results of these studies will be considered and the availability of any further data on survivorship of discarded fish investigated. Information on the mortality sustained by escapees seems to be equally sparse but initial results do suggest that survivorship of this element will be much higher than that of the discarded catch (See Section 8.2.2).

8.2.4.5. *Predicting benthic invertebrate mortality from fishing effort statistics*

In determining the benthic invertebrate mortality in a given area, it will be important to endeavour to include as much information as is possible on the gear and vessel specifications of the fleet operating in that area. The selectivity of the gear will affect the mortality of animals caught in the net and the type of groundgear will effect the mortality of invertebrates in the towpath. Early work on the difference in mortality caused by the different types of groundgear suggested that there was no sense in considering them separately (de Groot, 1984). More recent studies have however suggested that in just considering the disturbance caused by Otter trawls in comparison with Beam trawls, there are clear differences in both the selectivity of animals being caught in the net and the level of mortality of benthic invertebrates killed in the towpath (e.g. Philippart, 1998).

In a working paper presented at the workshop on fisheries disturbance, Cotter (2003) suggested that the selectivity of the 5 nominal gear categories operating on the NE coast of England was likely to be highly variable because of the frequent occurrence of small variants of mesh, square mesh panels, twine and footrope. In fact out of 275-discard observer trips on vessels operating in the NE coast whitefish (Cod and Whiting) and Nephrops fisheries, 180 different combinations of gear specification were found! This suggests that deriving indices of the disturbance caused by these gears is unlikely to be precise if the disturbance is only broken down to the level of the 5 specified gear categories. However, Cotter does observe that in inspecting the detailed data on variation in gear features, a feature expected to catch more small animals, e.g. small mesh size, is often confounded with features that would be expected to allow more escapees, e.g. square mesh panels. If this is the case then the variability in the actual ecological disturbance caused may not actually be so high.

To increase the potential for developing meaningful indices of ecological disturbance to benthic invertebrates from fishing it will be important to try and resolve the effort statistics with a number of key characteristics. These are the penetration depth and area of contact of the gear and the spatial

overlap of effort with the different community types. Depth of penetration of the gear is particularly important in predicting benthic invertebrate mortality. Clearly, the deeper the penetration of the gear, the more infaunal animals will be captured in the gear or mortally damaged in the towpath (Bergman & Hup, 1992). For some species it has been suggested that the depth of penetration may even affect the size selectivity of the animals killed, as certain species have a different depth distribution depending on size (See Section 3.3). The implications of this on the community level response of benthic invertebrates to fishing should be considered. Consultation of the reviews of the behaviour of different bottom fishing gears, will help to define a number of categories of gear, based on their penetration depth and the area of contact with the seafloor (For reviews see Watling & Norse, 1998; Auster & Langton, 1999; Johnson, 2002; Thrush & Dayton, 2002). It will also be important to account for the dependence of the actions of these gears on substrate type.

The spatial distribution of benthic invertebrates is known to be patchy and it is now thought that the distribution of effort can also be highly aggregated and heterogeneous (See Section 8.2.4.2). It will therefore also be particularly important, where possible, to resolve microscale distribution of fishing effort for the determination of benthic invertebrate disturbance indices. The resolution of data in order to account for the overlap in different communities with different levels of fishing effort will help to improve the precision of disturbance indices (Rijnsdorp et al., 1998; Piet et al., 2000). In trying to do this, the availability of data on the spatial distribution of particular community types will also be vitally important. Having microscale distribution of the effort data will provide little help if it is not known how communities are distributed within that area. A number of key papers that describe the distribution of characteristic infaunal and epifaunal invertebrate communities are available at the scale of the North Sea (Duineveld et al., 1991; Künitzer et al., 1992; Callaway et al., 2002). It will be important to consult all known sources of data on the distribution of both epifaunal and invertebrate communities before determining the disturbance indices.

There is also unequivocal evidence that the type of benthic substrate will affect the level of mortality of invertebrates in the towpath of the gear. This is partly because the level of penetration of groundgear will be affected by the type of substrate and also because there is a direct relationship between substrate type and the community composition of benthic invertebrates present in that area (Duineveld et al., 1991; Kaiser & Spencer, 1996b). When considering the invertebrate community response to fishing, substrate type will also have an important role. Communities in stable sediments, subject to low frequency natural physical disturbance have been shown to be less resilient to bottom trawling than communities subject to the same fishing regime in mobile sediment types (Kaiser & Spencer, 1996b).

8.2.5. Response of demersal fish and benthic invertebrate communities to fisheries disturbance

8.2.5.1. Implications of fisheries disturbance to demersal fish communities

It is certain that the community composition of demersal fish species in the North Sea has changed in the last 20-30 years (Greenstreet & Hall, 1996; Heessen & Daan, 1996; Greenstreet et al., 1999; Greenstreet & Rogers, 2000; Clark & Frid, 2001). Some of the evidence for why these changes have occurred infers a role for the effects of fisheries disturbance. This may be through the alteration in competition that has resulted from the removal of large numbers of targeted species or from changes in the availability of food resources due to the fisheries disturbance of benthic invertebrate communities. Jennings et al. (1998) examined the differential effects of fishing on

individual species with contrasting life histories. This work suggested that those species that decreased in abundance compared with their nearest relatives, matured later at a greater size, grew more slowly towards a greater maximum size and had lower rates of potential population increase. It was proposed that trends in community structure could be predicted from the differential responses of related species to fishing. Results agree with the prediction that fishing has greater effects on slow growing, larger species with later maturity and lower rates of potential population increase (Jennings et al., 1999a). More recent work concurs with this idea and suggests that the differential effects of fishing on species and populations with different life histories is a stronger and more universal indicator of fishing effects than changes in the mean trophic level (Jennings et al., 2002a).

Many benthic invertebrates are killed in the fishing process, either as targeted animals that are removed from the system, as discarded or escapee animals that are returned to the sea, or through contact with the gears in the towpath (See Section 3). There are a number of implications of this on demersal fish communities, some that may increase specific population growth rates and some that may decrease them. These must be considered in the modelling of the overall response of demersal fish communities to a fishing disturbance.

There is little published work on the response of demersal fish to the removal of benthic invertebrates from the system through the fishing process. It is possible that the removal of large numbers of a particular species in a fished area may affect some demersal fish communities through the decrease in competition for resources such as food and habitat. However, it is also conceivable that some of the targeted invertebrates, such as the smaller brown shrimp, may act as prey resources for other fish species and thus the overall community response is likely to be complicated. Most demersal fish species will be distributed over areas far greater than the area specifically targeted for an invertebrate resource. Thus the effect of the decrease in population size of an invertebrate species from one area will be inconsequential in comparison with other factors that structure the demersal fish community. It is also important to note that for some invertebrate fisheries the capture efficiency for the targeted species is notoriously low and so the overall affect on the invertebrate population may be too small to alter interactions with other benthic animals (Jenkins et al., 2001). The validity of these suggestions should be explored in developing a model of the demersal fish community response to invertebrate targeted fisheries.

There is far more information, however, on the response of scavenging and predatory demersal fish species to the increase in food resources left in the track of the fishing gear. As has been described in several of the earlier sections, considerable numbers of dead or dying fish and invertebrates are left in the wake of fishing vessels. These are the animals either discarded from the vessel or those that are killed or injured in the towpath of the gear. Groenewold & Fonds (2000) calculated that over 10% of the total annual secondary production of macrobenthic invertebrates becomes available, as damaged or displaced animals in the passage of a single beam trawl in the southern North Sea. There is clear evidence that mobile scavengers, including some demersal fish species, will actively move into a trawled area to take advantage of this increase in resources (Kaiser & Spencer, 1996a; Fonds & Groenewold, 2000). Highly mobile predators such as fish have been recorded to arrive at the fished area within 30 minutes of the disturbance occurring (Kaiser & Spencer, 1996a). A number of studies have investigated the ability of invertebrates to perform classic escape responses following either discarding from a vessel, or contact in the towpath of the gear. In most cases these studies have investigated target species and in all cases the escape response is greatly reduced, even when physical damage to the organisms is low, suggesting that specimens will be vulnerable to

greater levels of predation (Ramsay & Kaiser, 1998; Coffen-Smout & Rees, 1999; Wileman et al., 1999; Jenkins & Brand, 2001). For some species damage to individuals is so severe as to prohibit any escape response in over a quarter of all specimens in the track of the gear (Jenkins et al., 2001). The short term increase in food resources for scavenging and predatory demersal fish are likely to be significant but it will be important to try to quantify the relative importance at the community level over a wider area.

Even more difficult to quantify is the effect of the overall change in benthic invertebrate community, as a response to fisheries mortality, on the demersal fish community (See Section 5.2). Increased growth rates in some flatfishes have been linked to improved feeding conditions (Rijnsdorp & Vingerhoed, 2001; Rijnsdorp & Van Leeuwen, 1996) that are thought to be as a result of increases in the abundances of small polychaetes, over the same time period (Rijnsdorp & Van Leeuwen, 1996). These increases in the relative abundance of fast growing polychaetes in the benthic invertebrate community have been linked to sustained fisheries disturbance, but as pointed out by Jennings et al. (2002b), the analyses are complicated by increases in primary production over the same period.

The alteration of habitat structure in the towpath of the gear is not implicitly considered in this review but it is likely that loss of habitat important to the population growth of particular demersal fish species will have consequences at the community level. Habitats important to fish include spawning and nursery grounds, areas of specific feeding resources, areas of shelter from predators and areas of seabed that form part of a migration route (Benaka, 1999). The implications of reducing the availability of these habitats through the physical disturbance caused by fishing should be considered in developing a model of the demersal fish community level response to fisheries disturbance. Ideally, a change in habitat from one that is important to the fish community to one that is less so, or vice versa, should be incorporated in the disturbance index. This will be complicated at the community level however, as it is likely that different species will be associated with different types of habitat and will have different levels of dependence on particular habitat features.

8.2.5.2. *Implications of fisheries disturbance to benthic invertebrate communities*

Fisheries mortality of benthic invertebrates is largely an unknown quantity at the scale of the North Sea. However, it is clear that for some combinations of fishing gear and habitat type, mortality both in the gear and in the towpath of the gear is likely to be high for some components of the community (See Sections 8.2.3 and 8.2.4.5). The overall benthic invertebrate community level response to fisheries disturbance will depend on a number of factors. These include the absolute mortality following the passage of the gear, the effect of fisheries mortality to the demersal fish community and the effects of the alteration of habitat type following the passage of the gear. It is likely that the community level response will also vary dependent on the influence of a number of other drivers that structure the community. These include the level of local productivity, the local availability of propagules for immigration into the disturbed area and the influence of hydrography and climate.

In many areas of the North Sea, time series studies have inferred a role for fisheries in the long-term changes in community composition seen. This shift in composition could be a result of sustained fishing disturbance but the influence of climate and other anthropogenic drivers such as pollution and eutrophication cannot be discounted (Engel & Kvitek, 1998; Kaiser et al., 2000; Bergman & van Santbrink, 2000; Kröncke & Bergfeld, 2001). There is a suggestion that benthic invertebrate

communities have changed from those dominated by low productive, slowly reproducing organisms to quickly reproducing, opportunistic species. It is likely that the larger, slow-growing species (k-strategists) will be particularly vulnerable to sustained levels of mortality, whilst smaller individuals and species can endure higher mortality rates (Gilkensen et al., 1998). Clearly though, fishing may not be the only factor increasing mortality in these communities and the development of fisheries disturbance indices will help to elucidate how significant the mortality resulting from fishing is. Some studies have hypothesised that many of the large, high biomass infauna burrow below the depth that most bottom fishing gears will penetrate, confounding the suggestion that changes in size structure of benthic invertebrate communities may be a response to fisheries disturbance. Hall-Spencer et al. (2001) do however point out that although the large adults of some species do pass below the gear, the populations may still decline because there is reduced recruitment of juveniles that do live within the penetration depth of the gear. Jennings et al. (2001) suggest that the differential vulnerability of species to trawling leads to lower biomass and production of communities in heavily trawled areas and a dominance by smaller, faster growing individuals and species. Dinmore et al. (2003) describe Duplisea et al.'s (2002) size-based model that was used to assess the impacts of trawling on benthic production. For invertebrates in the range of 1µg to 80g (shell free wet weight), the model predicted that larger species could only survive in some fishing grounds because trawling disturbance was patchy.

Assessing the significance of the fisheries mortality of benthic invertebrates at the level of the community will be complicated by a number of factors. Even at the population level, there is evidence that low survivorship of particular species in the fishing process may not actually correspond with a significant change in population growth rates in the area. For example, although the discarding mortality of a number of key epifaunal species has been found to be high in some bottom trawling fisheries (e.g. see references in Bergmann et al., 2002), it is thought that the actual catchability of the gears for these species is very low (<10%, but between 10-70% for megafaunal epifauna in a beam trawl; Craeymeersch et al., 1998). This would mean that the effect at the local population level might actually be insignificant. This is even more likely when one considers that many of the invertebrate species that are caught as bycatch in trawl fisheries, are also the same scavenging species that benefit from the increase in resources that occurs following the passage of the gear (e.g. *Liocarcinus* spp., *Asterias rubens* and *Pagurus* spp.) (Bergmann, 2000). Even when species do not benefit directly from the increased food resources available following the passage of the gear, populations in many areas appear to be highly resilient to the levels of mortality sustained as bycatch. For example, Bergmann (2000) described the effect of bycatch mortality on populations of the brittlestar *Ophiura ophiura*. Although this echinoderm suffered 100% mortality in the bycatch process and on average made up 8% of the discarded catch, populations in the locality were highly abundant and it is suggested that the reproductive resilience of this species allows it to sustain high levels of mortality.

To further complicate the community level response of benthic invertebrates to fishing, there are also a number of effects of fisheries disturbance that may lead to increases in populations growth rates for some species. These include the increase in food resources for scavengers and the potential decrease in predation rates by fish that are removed in the fishing process. Again however, the signals from these changes may not be as straightforward as could be expected. Frid et al., (1999) actually found there to have been an increase in predation on the benthos, at the same time as an overall decrease in demersal fish biomass in the North Sea. They suggest that fishing has removed greater quantities of higher biomass gadoids, whose diet is principally piscivorous, allowing for

increased population growth rates in some flatfish and young gadoids, which do prey on benthic invertebrates.

There have been a number of studies on the response of benthic invertebrate scavengers to the availability of moribund material in the towpath of the gear and the deposition of discards on the seafloor following the release of discarded bycatch over the side of the vessel. Some of this material will float and large quantities of discarded fish are taken by scavenging seabirds (Hudson & Furness, 1988; Garthe et al., 1996). The remaining discards, except for a small amount taken by fish and marine mammals in the water column, will however fall to the seafloor. Although some discards will survive, many will be dead already or will have suffered high levels of physical stress and thus will be vulnerable to predation from demersal scavengers. As described in Section 8.2.5.1, a study of the moribund invertebrate material left in the trawl track of a southern North Sea beam trawl fishery was comparable to greater than 10% of secondary macrofaunal production available in the area (Groenewold & Fonds, 2000).

Although there have been some inferences to a link between increased population sizes of some scavenging seabirds and increased availability of fisheries discards (e.g. Furness, 1984), it is much more difficult to draw the same conclusions for populations of benthic invertebrates. In the first place, we do not have nearly enough information on changes in population structure of any benthic invertebrates. The only information we really have is from studies that have examined the abundance and density of benthic scavengers in the vicinity of fisheries induced moribund material (e.g. Kaiser & Spencer, 1996a; Ramsay et al., 1997; Hall-Spencer et al., 2001). The findings of these studies suggest that aggregations of scavenging invertebrate species do occur around areas of fisheries disturbance. The most work has been done on the larger epibenthic species that are more easily monitored through video and still camera exposure. These include Crustaceans such as *Pagurus bernhardus* (hermit crab), *Liocarcinus depurator* and *L. holsatus* (swimming crabs) and *Cancer pagurus* (the edible crab); the starfish *Asterias rubens* and the whelk *Buccinum undatum* (Hall-Spencer et al., 2001). Ramsay et al. (1997) also tried to account for the aggregation of smaller invertebrates by using baited traps. They found that a number of amphipods, mysids and isopods were caught in the baited traps, but that more work would be needed to establish the significance of this increase in food resources to the smaller animals of the invertebrate community.

Although this review does not implicitly cover the alteration of habitat that results from demersal fishing, there is evidence in the literature to suggest that there will be an overall community level response of benthic invertebrates to this kind of fisheries disturbance (For reviews see, Watling & Norse, 1998; Johnson, 2002; Thrush & Dayton, 2002). The physical alterations caused by the passing of the gear will in most cases change heterogeneity of the sediment surface, alter the texture (particle size composition) of the sediments and change the structure available to biota as habitat. Since the distribution of most benthic macrofaunal species in the North Sea is related directly to sediment particle-size composition and organic content (Duineveld et al., 1991), the physical disturbance associated with fishing effects will inevitably have consequences on the structure of the benthic invertebrate community. Although alteration of habitat may affect particular life stages of demersal fish (See Section 8.2.5.1 above), benthic invertebrates have close associations with the benthic habitat throughout their lifecycle. Thus, it is perhaps even more important that the implications of habitat change are included in the fisheries disturbance indices derived for benthic invertebrates.

8.2.6. Final comments

Fishing effects the fish community directly through the removal of fish from the community. This fishing induced mortality is not evenly distributed across all individuals within the community. Fishing is a highly selective activity. Over 220 species have been recorded in the North Sea (Yang 1982a; 1982b), but less than 20 of these are specifically targeted by fishermen, with perhaps a further 20 that would be landed if incidentally taken in the bycatch. As part of the management process, minimum landing sizes are set for most commercial species, thereby imposing a strong degree of size selection that is further enhanced by the economic premium accorded at the market place to larger sized fish. This selective mortality can reduce the abundance of prey to predators, reduce natural (predation) mortality on smaller species and individuals, and affect the outcome of competitive interactions. Alteration of the habitat can affect recruitment rates and generally alter the “niche-spectrum” of an area (Auster et al., 1996; 1997; Auster & Langton 1999). Generally benthic communities are affected by fishing through a very similar set of processes, with one possible major difference; except where benthic invertebrates are the target of a particular fishing activity (eg on *Nephrops* fishing grounds), benthic invertebrates will not be specifically targeted by fisheries. Thus there will be no overt targeting of particular species and size classes, so differential mortality across individuals within the community might be expected to be less of an issue. As a result, the consequential effects of fishing might not be so profound among benthic invertebrate communities. However, the evidence presented in this review strongly suggests that this is not likely to be the case. Although fishing mortality might not be explicitly directed towards certain species and size classes, nevertheless, individuals and species within benthic communities vary considerably in their vulnerability to trawls gears (Collie et al 2000; Kaiser et al 2006). Fishing induced mortality is therefore not evenly distributed across all individuals in invertebrate communities, and the same processes that induce change within fish communities almost certainly operate to cause change in benthic invertebrate communities.

8.3. Compilation of international landings databases (MAFCONS participants)

Landings data provide a direct measure of the impact of fishing on commercial fish in the North Sea. Landings are sampled regularly by each country prosecuting a fishery under the Common Fisheries Policy (CFP). These data are essential to the annual stock assessment process where they contribute directly to the estimates of annual fishing mortality of each stock (ICES 2005). Thus landings data provide a direct measure of the ecological disturbance suffered by the North Sea fish community as a result of fishing activity. Generally stock assessments are made at the spatial scale of the stocks concerned, ie, ICES area IV and IIIa (The North Sea, Kattegat and Skagerrak) and total landings data for each species from this whole area are readily available (ICES 2005). However, for the purpose of spatial testing of Huston’s Dynamic Equilibrium Model, spatial variation in the disturbance inflicted on the fish community were required. To address this issue, an international landings database, with data available at the ICES rectangle scale, was compiled.

Data were supplied as the number of tonnes of each of the main commercial demersal species (cod, haddock, whiting, saithe, plaice, sole and *Nephrops*) caught by demersal fishing gears from each ICES statistical rectangle in each year. Data were made available by each of the participating MAFCONS countries except Belgium (UK {England, Wales and Northern Ireland [E.W.N.I.]}, UK {Scotland [S]}, The Netherlands, Germany and Norway. The resolution of fishing gear information differed between countries. For example, UK (E.W.N.I.) and UK (S) provided data for 11 and 12

different gear codes respectively, while other countries tended to provide information for major gear categories (eg otter trawl and beam trawl). Data therefore need to be aggregated to “lowest common categories”. Despite this, taking account of ancillary information regarding TACs and total North Sea landings for each country (ICES 2005, Chapter 12), meant that data could be analysed for six different major fishing gear categories; human consumption fish otter trawl, industrial fish otter trawl, *Nephrops* otter trawl, other invertebrate otter trawl, beam trawl and seine netting. For example, Norway, UK (E.W.NI.) and UK (S) all provided information that allowed human consumption fish, *Nephrops* and other invertebrate otter trawls to be separated, while this was not possible for Germany and the Netherlands. However, these other two countries do not have significant TACs for *Nephrops* and land very little of this species. Similarly, neither Germany or the Netherlands prosecute a significant industrial fishery. Consequently, landings by otter trawl for these two countries were assumed to be all by human consumption fish otter trawl. On the basis of this, it was therefore possible to separate the four otter trawl categories.

Before carrying out detailed spatial analysis of the landings data provided by each of the institutes collaborating in the MAFCONS project, annual total landings were summed and compared with total landings from the five countries involved presented by ICES (2005) as part of the stock assessment process (Figure 8.3.1). Close agreement was observed for all species with the exception perhaps of plaice and saithe. For plaice a tight linear relationship between landings reported to ACFM for the five MAFCONS partners and landings data recorded in the MAFCONS landings database was observed ($r^2=0.98$), but the regression coefficient, at $b=1.403$, was higher than the expected value of one, suggesting a constant ratio bias. Such systematic variation suggests the intervention of a constant procedural process, such as how landings are assigned to different countries during the stock assessments to take account of vessels operating under different countries flags. Thus, for example, much of the Belgian beam trawl fleet has been bought up by Dutch operators. Landings and effort for the vessels concerned are included in the MAFCONS database, but for the purposes of the stock assessments, landing by these vessels are still counted against the Belgian quota. Quota swapping is also known to take place and failure to take this into account could also explain the apparent discrepancies indicated in figure 8.3.1. Generally the analysis illustrated in Figure 8.3.1 suggests a close correspondence between the data analysed in section 8.3.1 to derive spatial patterns and the data used in the stock assessments.

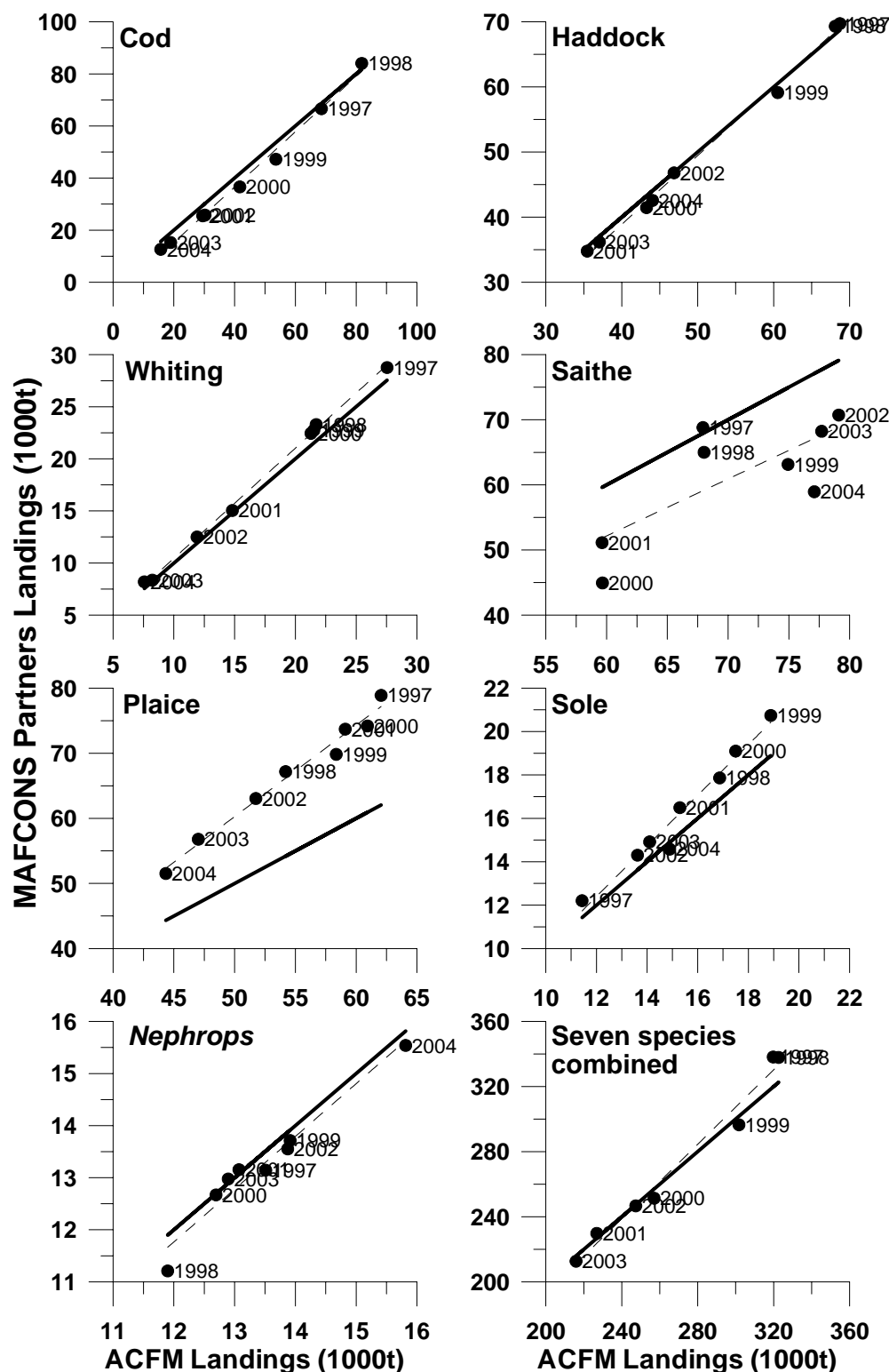


Figure 8.3.1. Correspondence between the landings data analysed to determine spatial patterns in landings as part of the MAFCONS project and the data reported to ICES for the stock assessments. Plots show total annual landings of each species, and all seven species combined, derived from the ACFM report (ICES 2005) summarising the ICES advice following the 2005 stock assessment process and annual summed landings data derived from the MAFCONS database for the five countries involved in the MAFCONS project. Solid lines indicate relationship expected for perfect agreement and dashed lines show linear fits to the data.

8.3.1. Temporal variation in North Sea international landings

Over the period 1997 to 2004, landings of cod, haddock, whiting, saithe, plaice, sole and Nephrops by the countries contributing to the MAFCONS landing database (UK E.W.NI., UK S., Germany, Norway and the Netherlands) declined by approximately 40% (Figure 8.3.1.1.). Landings data reported by ACFM (ICES 2005) for the entire North Sea also indicated a reduction, although the proportional change, at approximately 33%, was not so high (Figure 8.3.1.1.). Landings of cod, haddock, whiting and plaice declined over this period. Cod showed the most substantial reduction (81%) closely followed by whiting (71%). Landings data provided by the MAFCONS participants represented a high proportion of total North Sea landings, although this proportion declined significantly ($r^2=0.706$) over the period, from around 78% to 72%. This trend in total landings of all seven species combined does not convey the whole story however. The seven species examined fell into two distinct groups. The first, consisting of haddock, whiting, plaice, sole and Nephrops were largely landed by the MAFCONS participating countries, so that landings by these countries made up between 80% and 95% of total North Sea landings. The fraction of the total North Sea cod and saithe landings landed by the MAFCONS countries was substantially lower at between 55% and 65%.

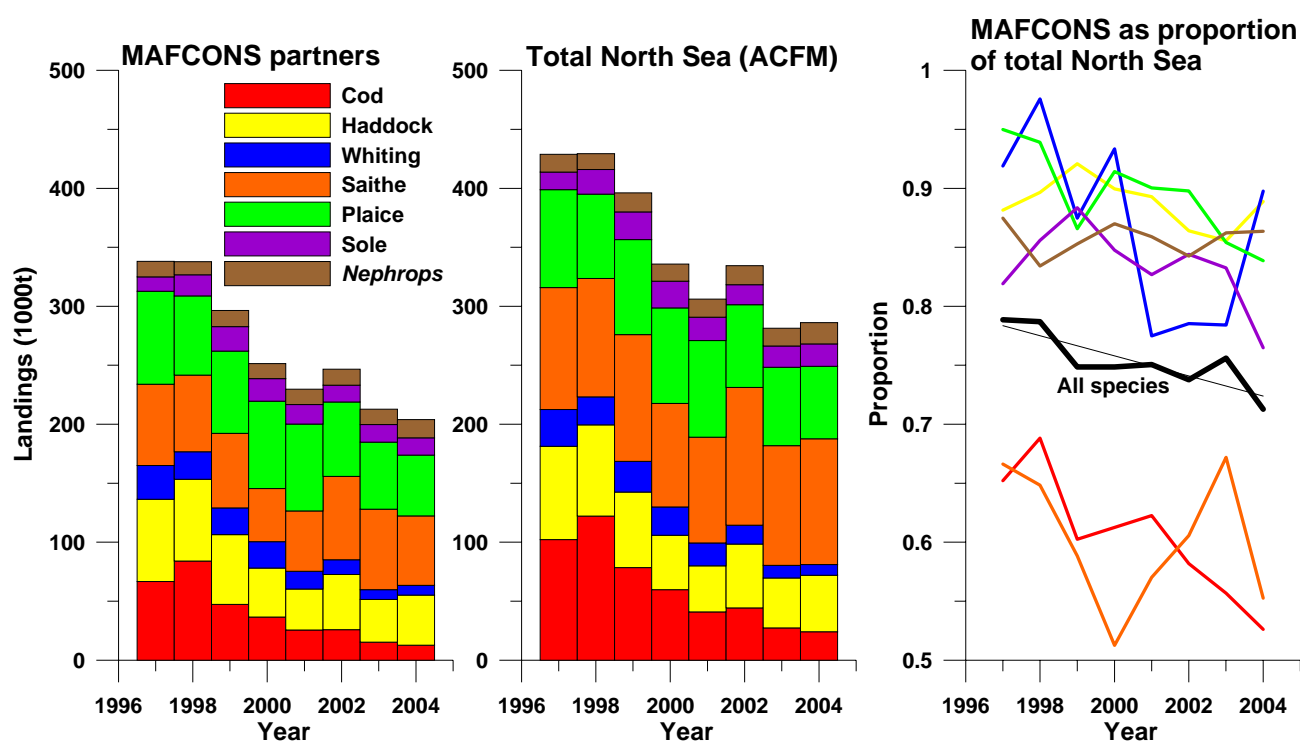


Figure 8.3.1.1. Trends in the combined landings of cod, haddock, whiting, saithe, plaice, sole, and Nephrops landed by the countries participating in the MAFCONS project (left panel), and in total North Sea landings reported by ACFM (central panel). Trends in the proportion of total North Sea landings taken by the MAFCONS countries are shown in the panel to the right.

The ACFM landings data were examined to determine the importance of each species to all countries operating demersal fisheries in the North Sea. This was done primarily to determine the relative importance of each species to countries outside the MAFCONS partnership, and for which access to landings and effort data were limited. Cod and Saithe were the two species with the largest

fraction of total landings taken by countries outside the MAFCONS partnership. Denmark, followed by Belgium and France, were the principal exploiters of cod and France, followed by Denmark, were the principal exploiters of saithe. Of the remaining species for which by far the greater fraction of total landings were taken by countries within the MAFCONS consortium, haddock were also taken in significant quantities by Denmark, whiting by France, plaice by Denmark and Belgium, sole by Belgium, and *Nephrops* by Denmark and Belgium (Figure 8.3.1.2).

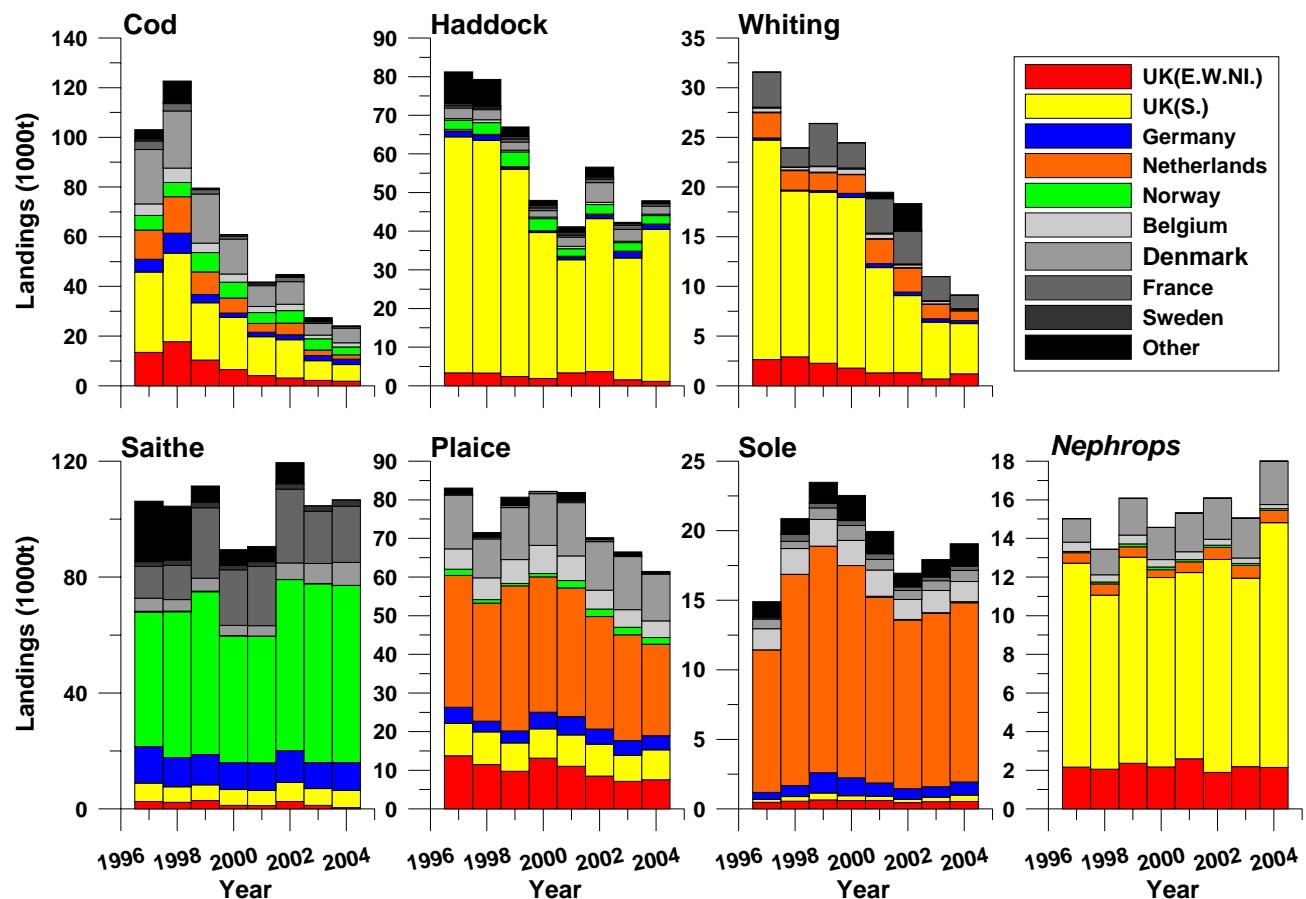


Figure 8.3.1.2. Trends in the landings of cod, haddock, whiting, saithe, plaice, sole, and Nephrops taken by the major countries operating demersal fisheries in the North Sea (ACFM data in ICES (2005)).

The four gadoid species were predominantly taken by otter trawl directed at fish for human consumption (Figure 8.3.1.3). This gear accounted for almost all the saithe landed. Seine gear also accounted for a significant fraction of the cod, haddock, and whiting landed and a significant proportion of cod and whiting landed were caught in beam trawls. Nearly all the plaice and sole landed were caught in beam trawls, although significant numbers of plaice were also caught in otter trawls directed at fish for human consumption (Figure 8.3.1.3). Between 25% and 33% of the *Nephrops* landed were taken in otter trawls directed at fish for human consumption, but the bulk of *Nephrops* landed were caught in otter trawl directed at them (Figure 8.3.1.3).

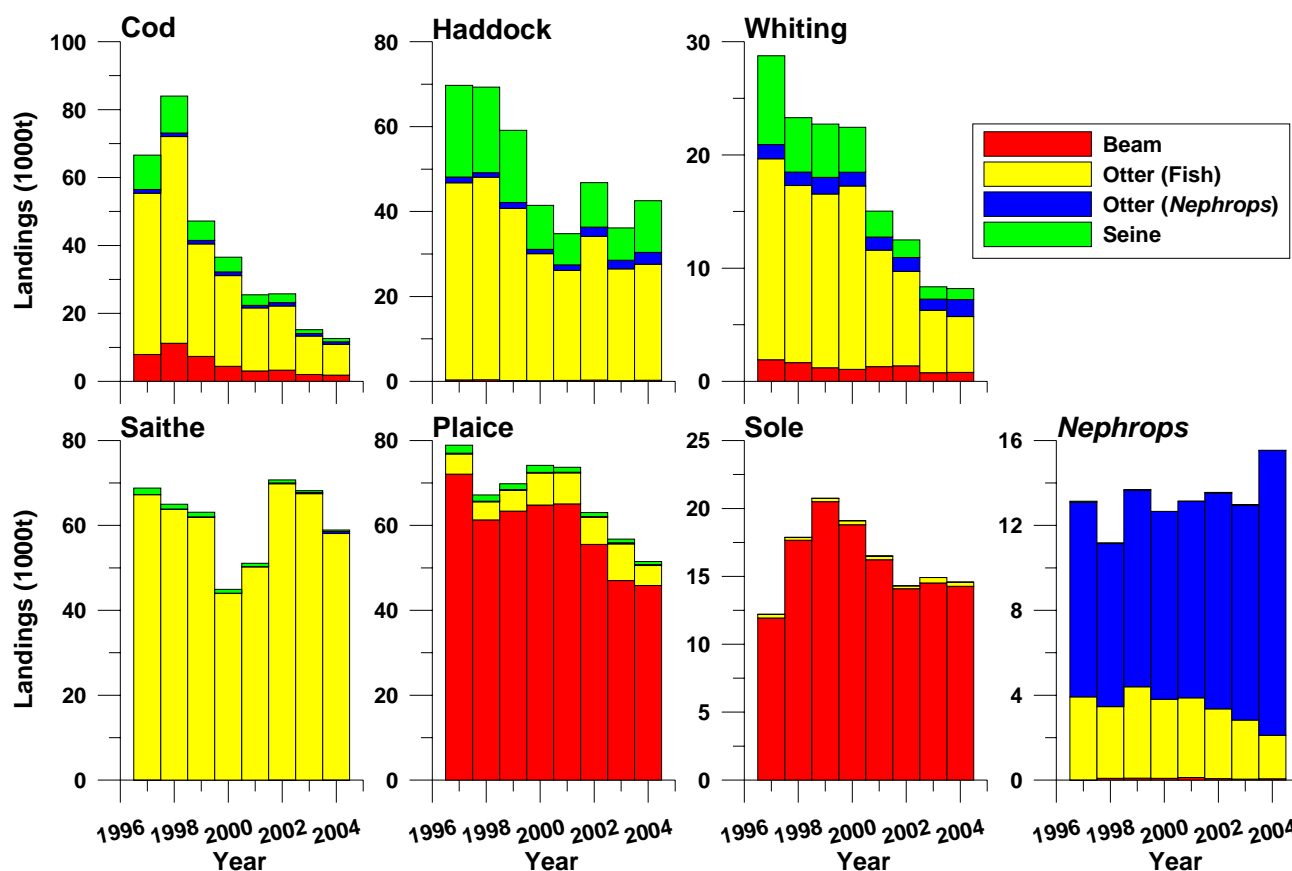


Figure 8.3.1.3. Trends in the landings of cod, haddock, whiting, saithe, plaice, sole, and *Nephrops* landed by the countries participating in the MAFCONS project (MAFCONS landings database) indicating the amounts caught in four major gear categories; beam trawl, otter trawl directed at fish for human consumption, otter trawl directed at *Nephrops* and seine gear.

8.3.2. Spatial variation in North Sea international landings

8.3.2.1. *Total landings (MAFCONS participants)*

Distributions of the total landings of cod, haddock, whiting, saithe, plaice, sole, and *Nephrops* in each year over the period 1997 to 2004 are shown in Figures 8.3.2.1.1 to 8.3.2.1.7. At the start of the period, cod were landed from most ICES rectangles across the North Sea at levels of 50t.yr⁻¹ or more. However, the major cod producing regions were located in the northeastern North Sea and in the southwest. By 2004, significant cod landings were mainly restricted to these two “hotspot” locations (Figure 8.3.2.1.1). Throughout the time period, haddock were rarely landed from the southeastern North Sea and the major haddock producing ICES rectangles were located in the northern and northwestern North Sea (Figure 8.3.2.1.2). Whiting were primarily landed from two “hotspot” regions, one situated in the northern North Sea and the second in the southwestern North Sea (Figure 8.3.2.1.3). Few saithe were landed from the southern North Sea. Major saithe landings were taken from the extreme northern and northeastern North Sea (Figure 8.3.2.1.4). Both plaice and sole were primarily landed from the southern North Sea, but whilst sole landings were almost entirely restricted to this area (Figure 8.3.2.1.5), plaice were also landed in significant quantities

from ICES rectangles in the northern North Sea (Figure 8.3.2.1.6). *Nephrops* were principally landed from ICES rectangles in the eastern and northern North Sea, but in recent years a few ICES rectangles in the southeastern North Sea have also started to produce significant landings of *Nephrops* (Figure 8.3.2.1.7).

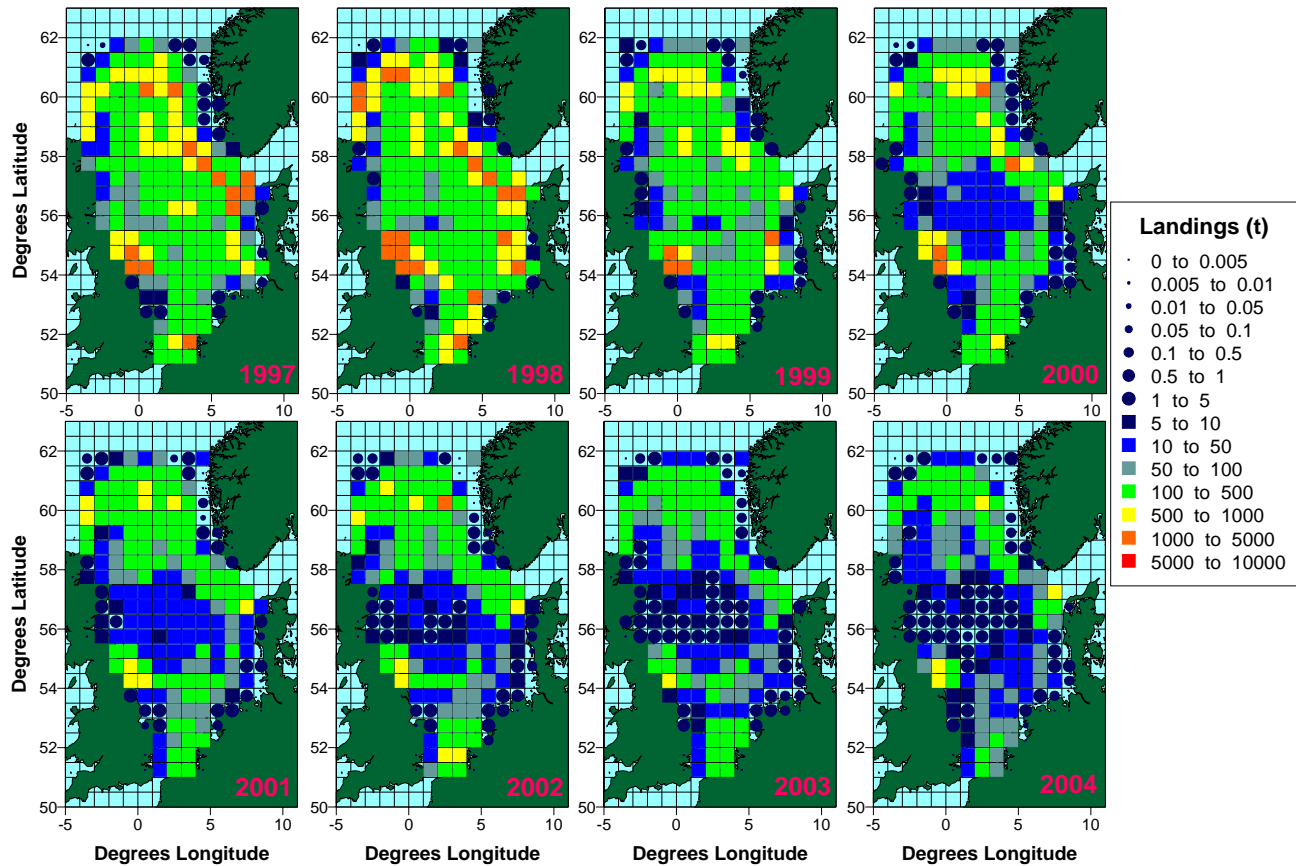


Figure 8.3.2.1.1. Spatial distributions of cod landings across all fishing gears by countries participating in the MAFCONS project in each of the years between 1997 and 2004.

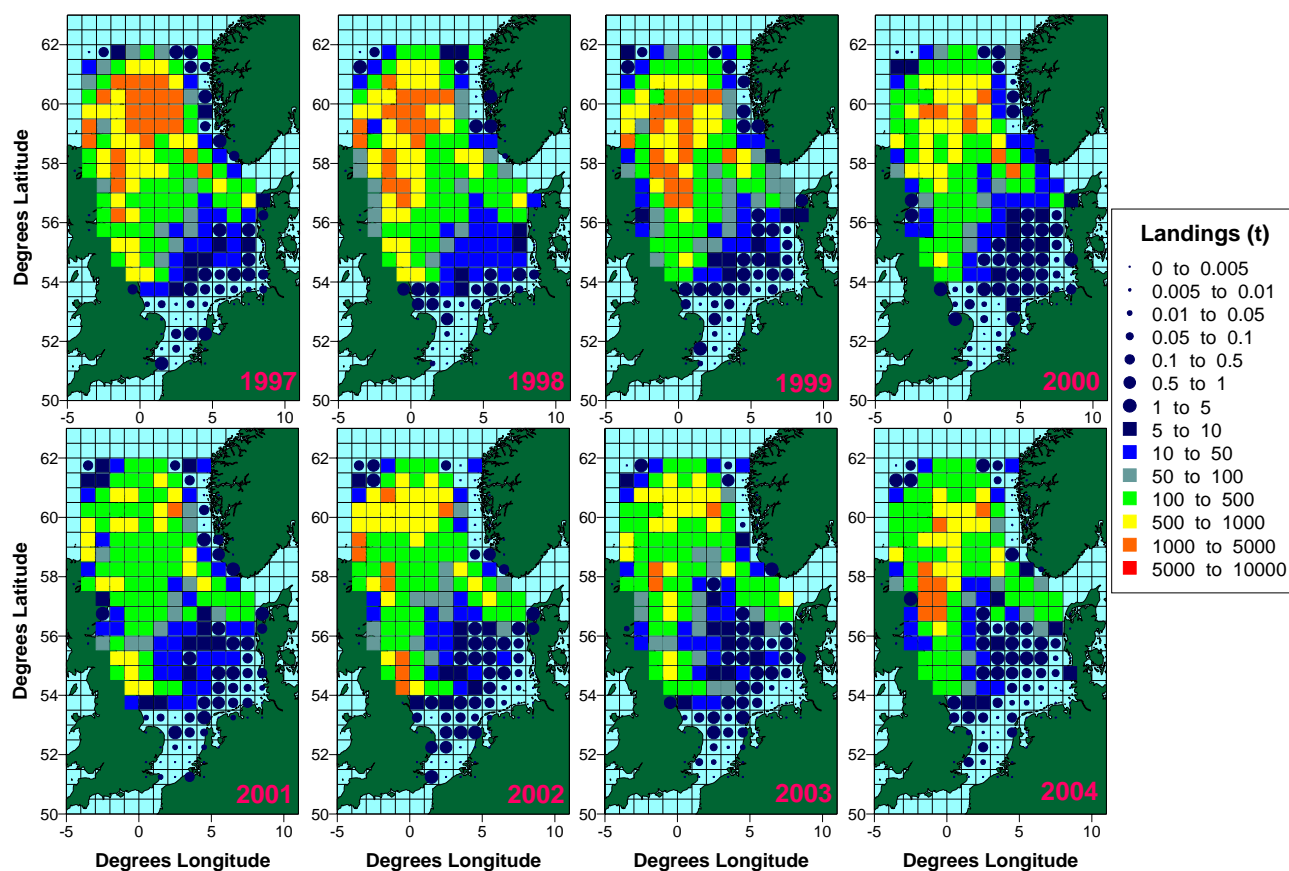


Figure 8.3.2.1.2. Spatial distributions of haddock landings across all fishing gears by countries participating in the MAFCONS project in each of the years between 1997 and 2004.

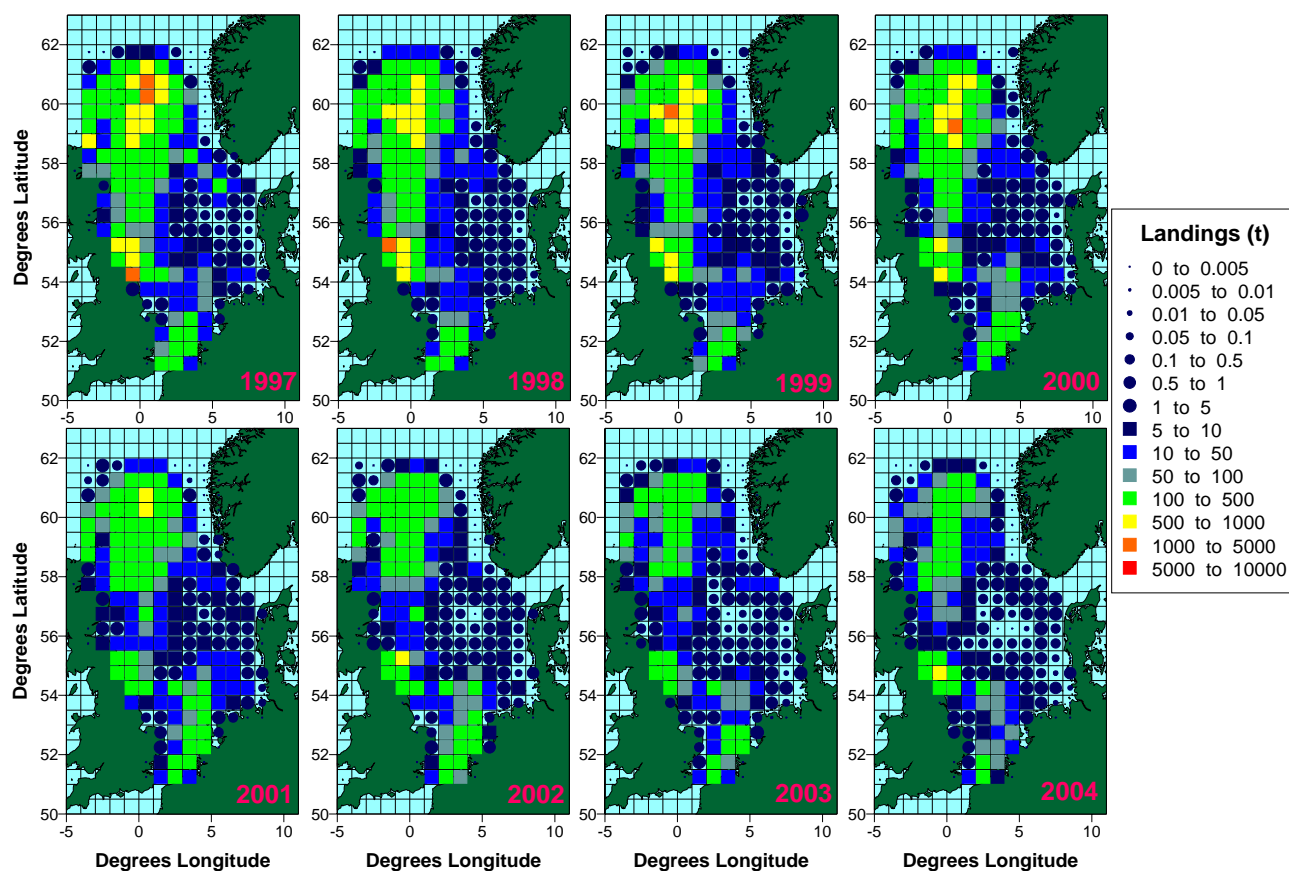


Figure 8.3.2.1.3. Spatial distributions of whiting landings across all fishing gears by countries participating in the MAFCONS project in each of the years between 1997 and 2004.

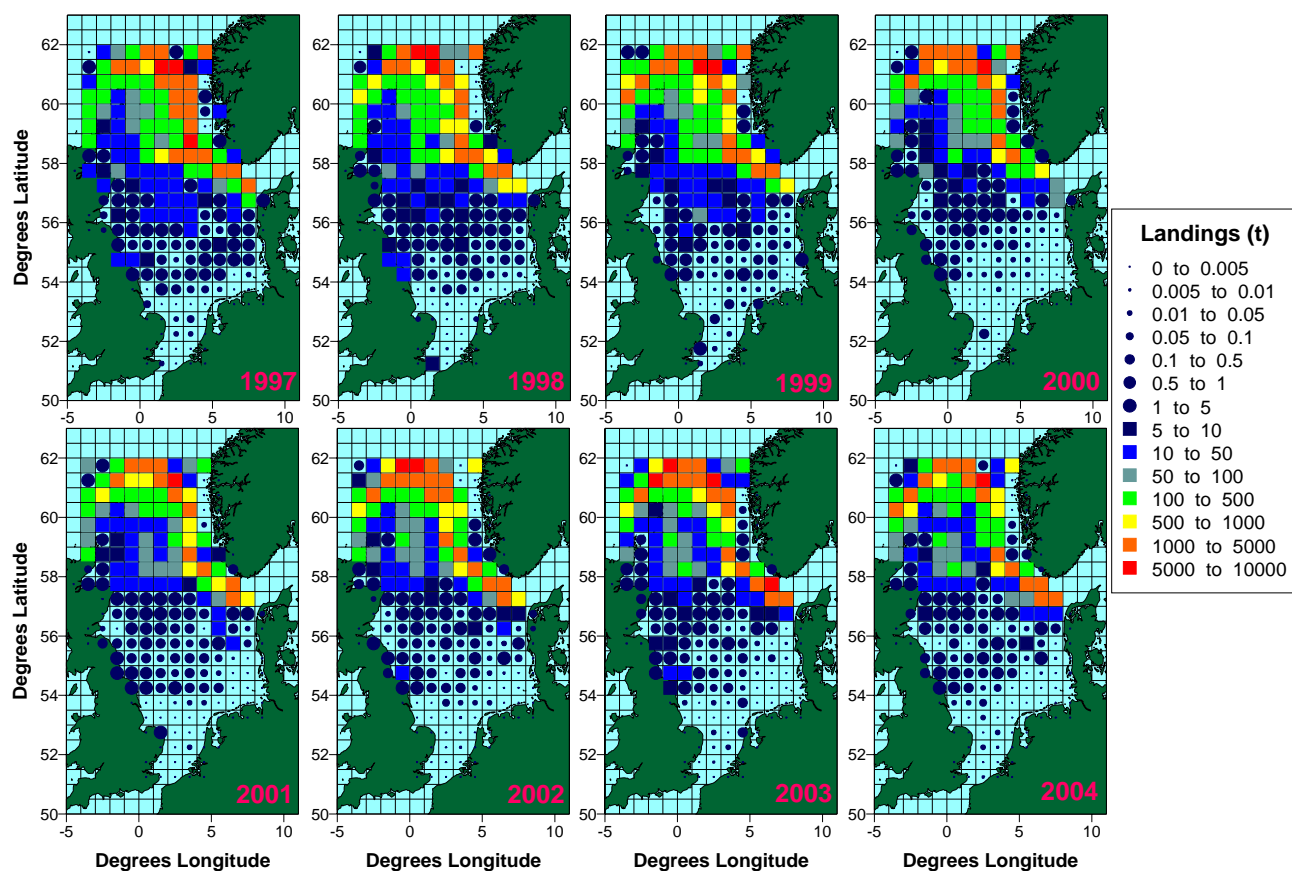


Figure 8.3.2.1.4. Spatial distributions of saithe landings across all fishing gears by countries participating in the MAFCONS project in each of the years between 1997 and 2004.

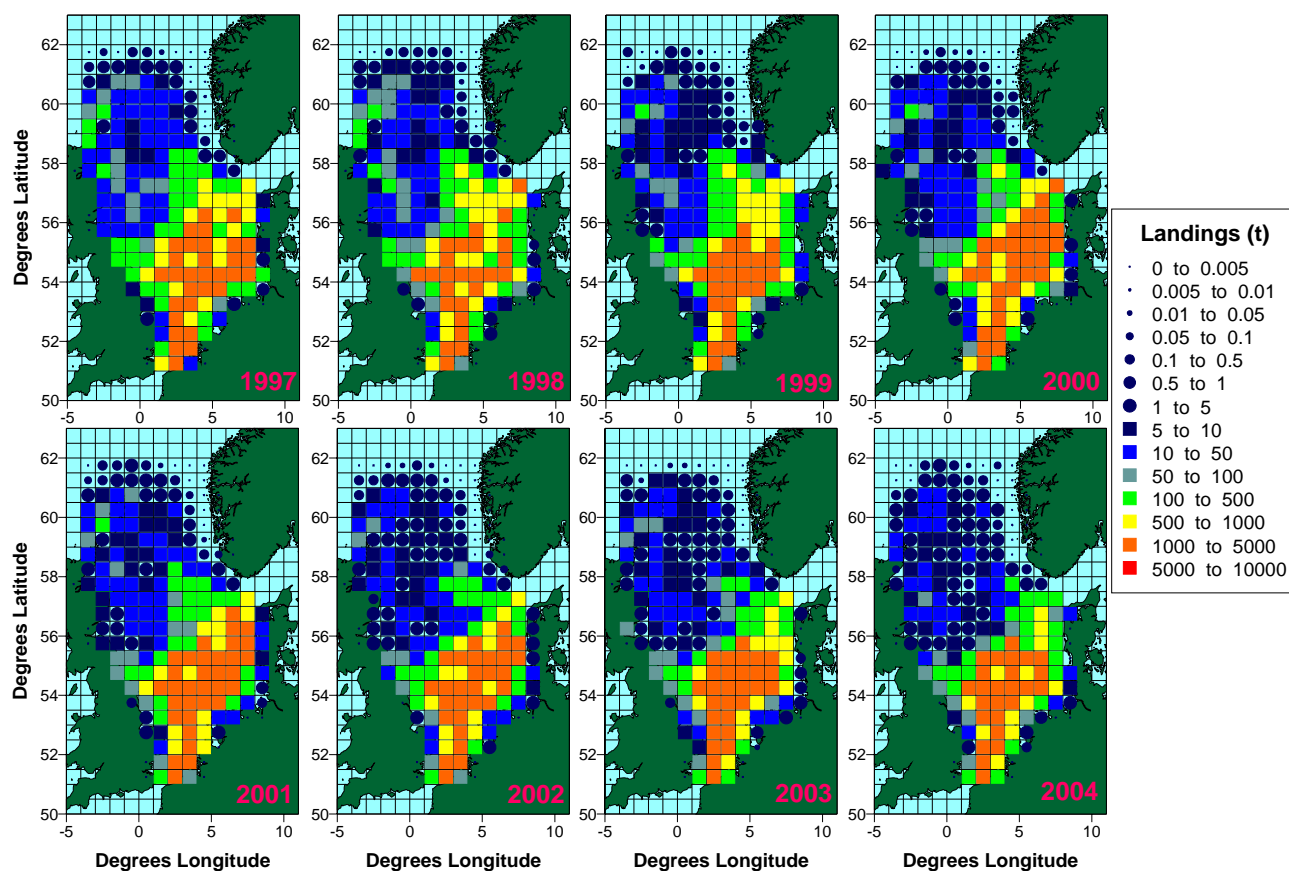


Figure 8.3.2.1.5. Spatial distributions of plaice landings across all fishing gears by countries participating in the MAFCONS project in each of the years between 1997 and 2004.

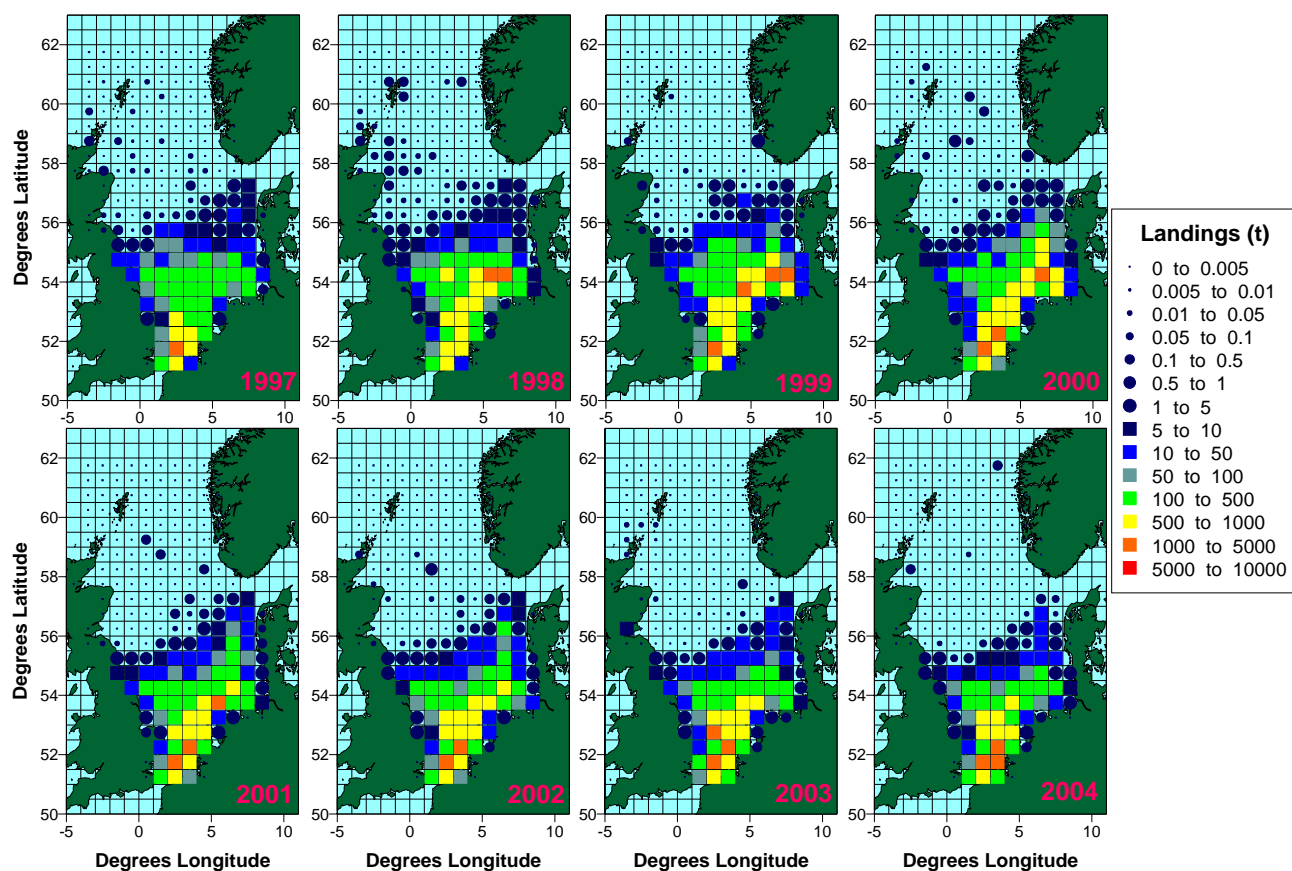


Figure 8.3.2.1.6. Spatial distributions of sole landings across all fishing gears by countries participating in the MAFCONS project in each of the years between 1997 and 2004.

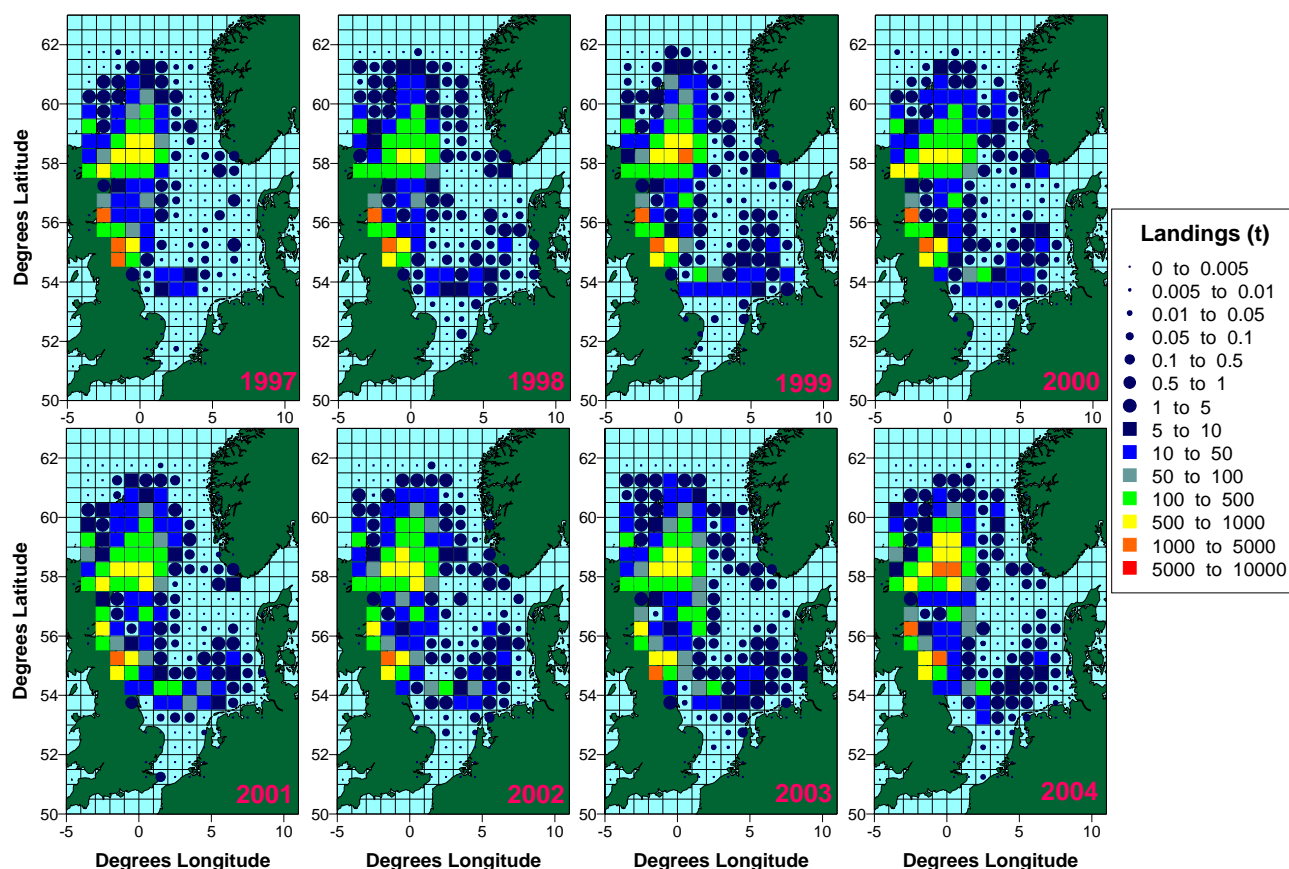


Figure 8.3.2.1.7. Spatial distributions of *Nephrops* landings across all fishing gears by countries participating in the MAFCONS project in each of the years between 1997 and 2004.

8.3.2.2. Landings by main gear category (MAFCONS participants)

Because of the temporal trends in total landings illustrated in Figures 8.3.1.1 to 8.3.1.3, and the associated changes over time in the spatial distributions of the landings of each species (Figures 8.3.2.1.1 to 8.3.2.1.7), average annual landings of each species by main gear categories were examined for two separate time periods, 1997 to 2000 and 2001 to 2004. Although differences in absolute values were apparent, driven by the temporal variation in total landings (Figure 8.3.1.1), essentially distributions of landings of each species by each of the four main gear categories examined were the same in each of the two time periods (Figures 8.3.2.2.1 to 8.3.2.2.7). Cod were taken in significant quantities in all four gear categories and landings distributions by gear (Figure 8.3.2.2.1) reflected both the distribution of total cod landings (section 8.3.2.1) and of fishing effort by gear (section 8.4.4.). The same was true for haddock (Figure 8.3.2.2.2), whiting (Figure 8.3.2.2.3) and plaice (Figure 8.3.2.2.5). Saithe were primarily taken in otter trawl directed at human consumption fish and seine gear (Figure 8.3.2.2.4), while sole were mainly landed from beam trawlers and otter trawlers fishing for human consumption fish (Figure 8.3.2.2.6). Again landings by gear for these two species reflected both the distributions of total landings and the distributions of fishing effort of the appropriate gear. Two gears, otter trawl directed at *Nephrops* and otter trawl directed at fish for human consumption, accounted for the bulk of the *Nephrops* landed (Figure 8.3.2.2.7). Landings by both these two gears most reflected the distribution of total *Nephrops* landings, which was similar to the distribution of effort of otter trawl directed at *Nephrops*.

Nephrops landings from otter trawl directed at fish for human consumption bore little relationship to the distribution of effort by this gear, simply because of the strong dependence of this species to specific seabed habitat characteristics.

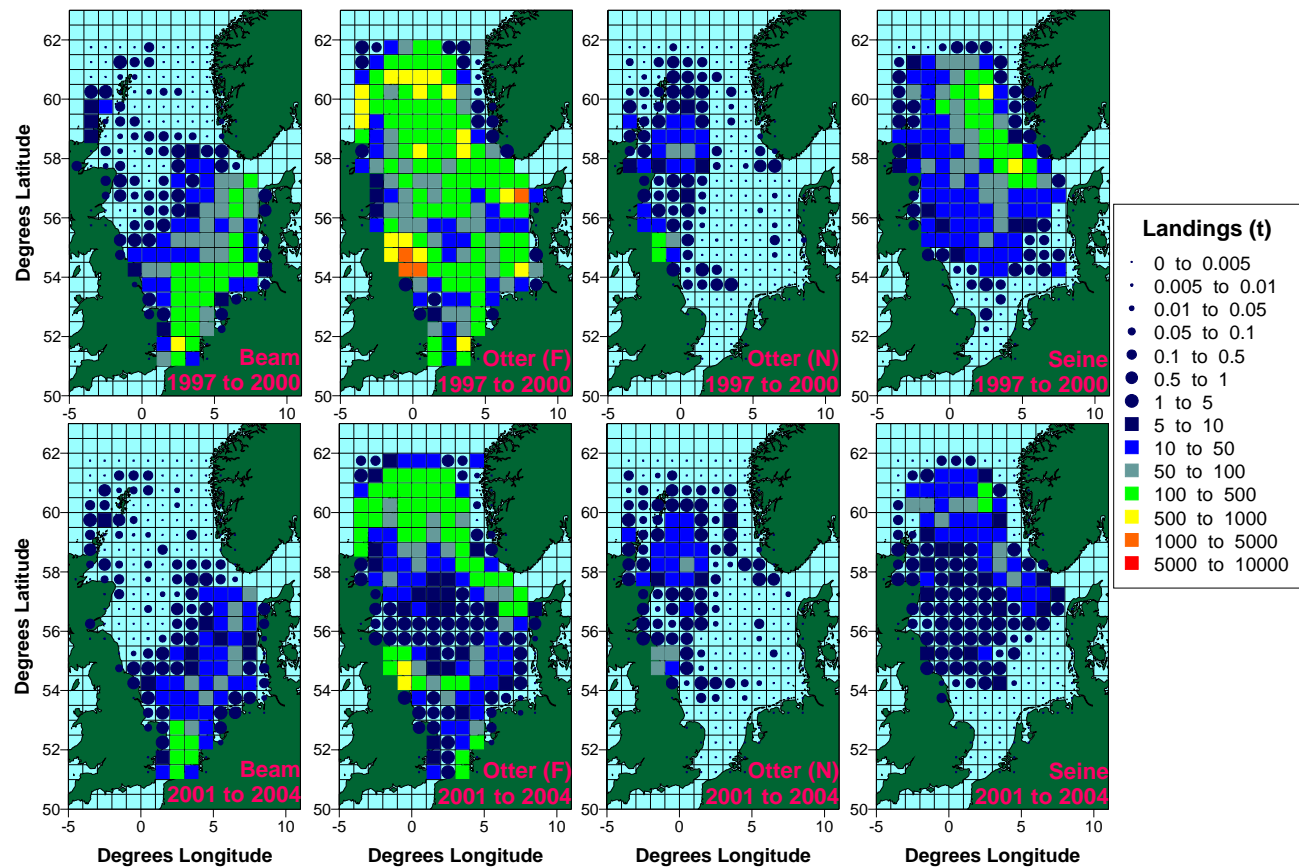


Figure 8.3.2.2.1. Spatial distributions in the average annual landings of cod taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear, in two four year periods, 1997 to 2000 and 2001 to 2004.

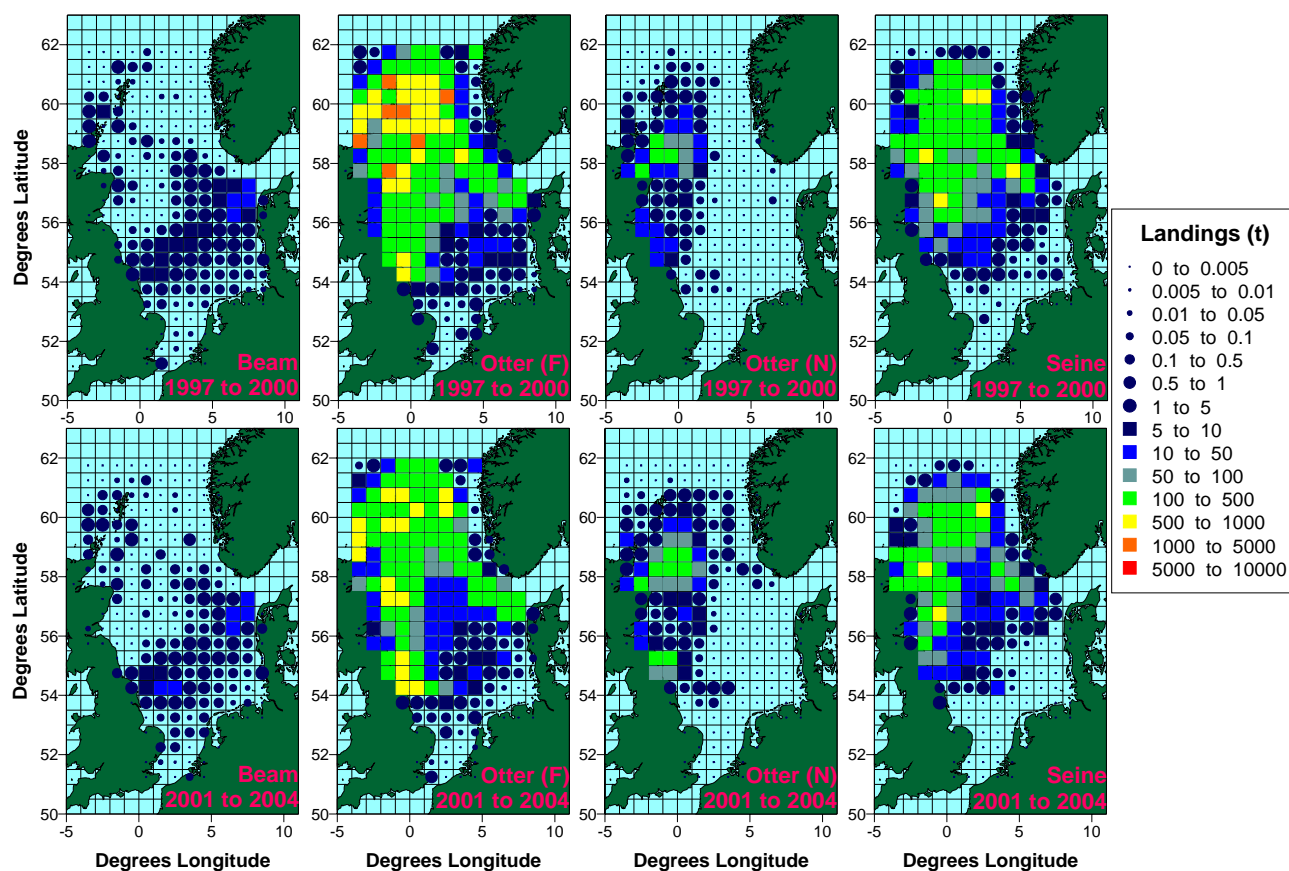


Figure 8.3.2.2.2. Spatial distributions in the average annual landings of haddock taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear, in two four year periods, 1997 to 2000 and 2001 to 2004.

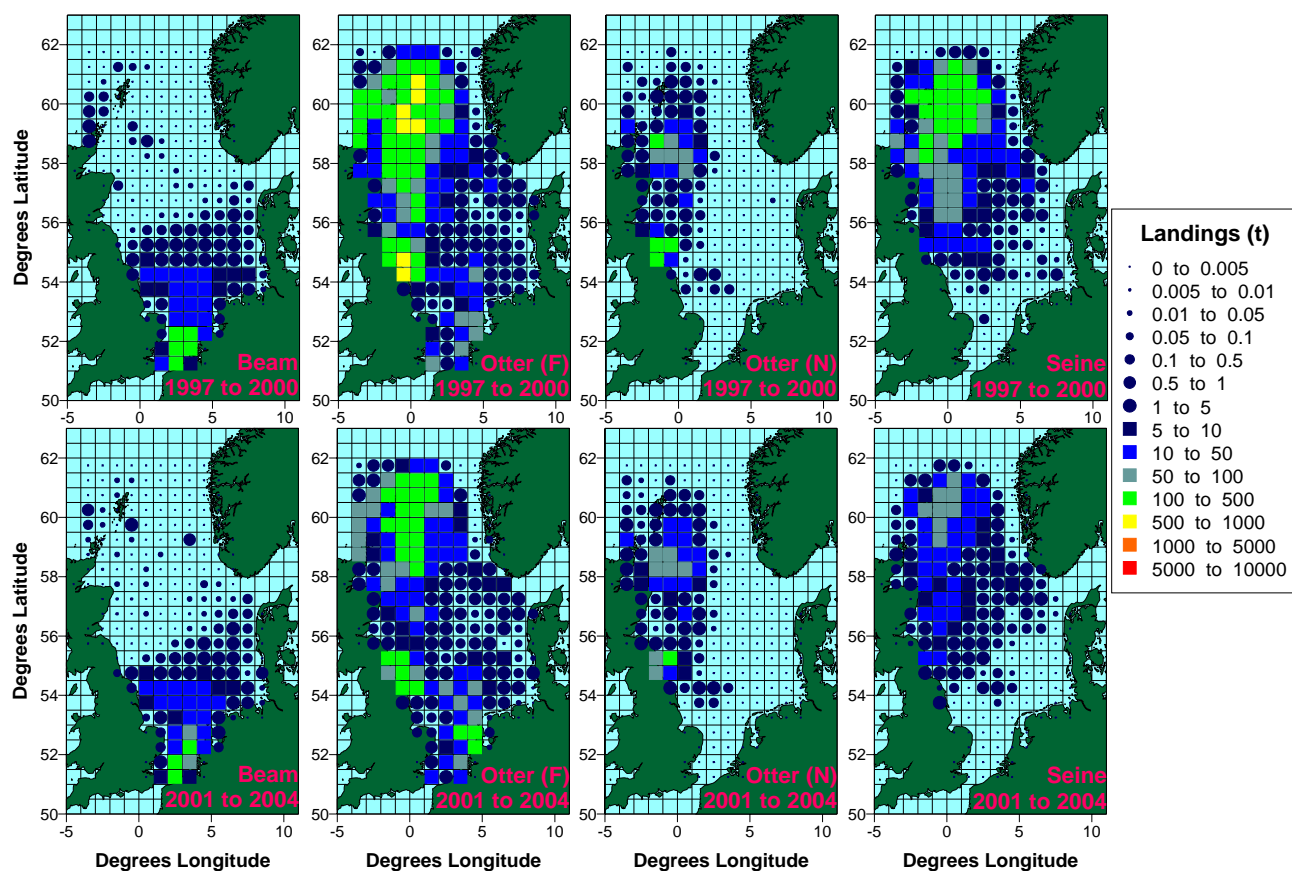


Figure 8.3.2.2.3. Spatial distributions in the average annual landings of whiting taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear, in two four year periods, 1997 to 2000 and 2001 to 2004.

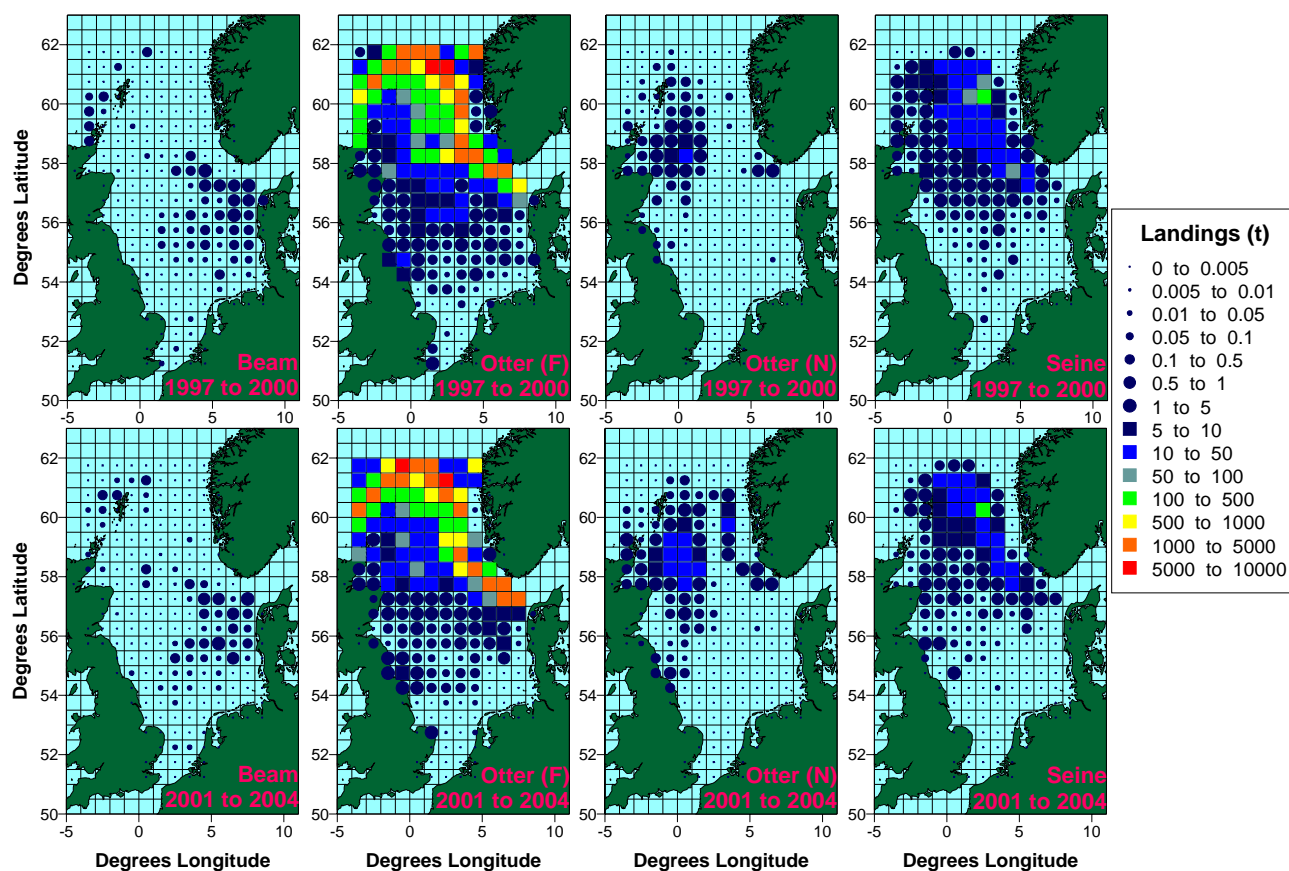


Figure 8.3.2.2.4. Spatial distributions in the average annual landings of saithe taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear, in two four year periods, 1997 to 2000 and 2001 to 2004.

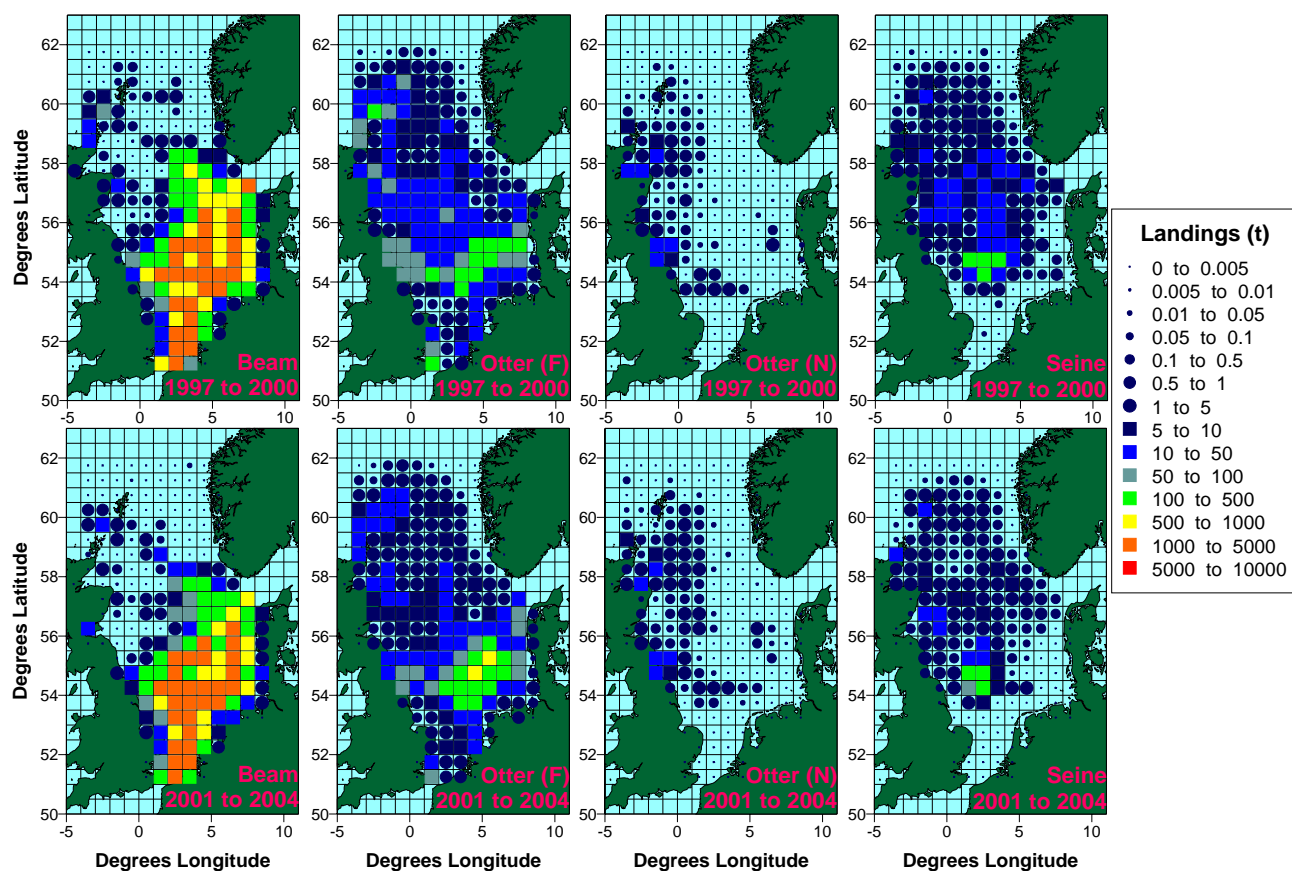


Figure 8.3.2.2.5. Spatial distributions in the average annual landings of plaice taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear, in two four year periods, 1997 to 2000 and 2001 to 2004.

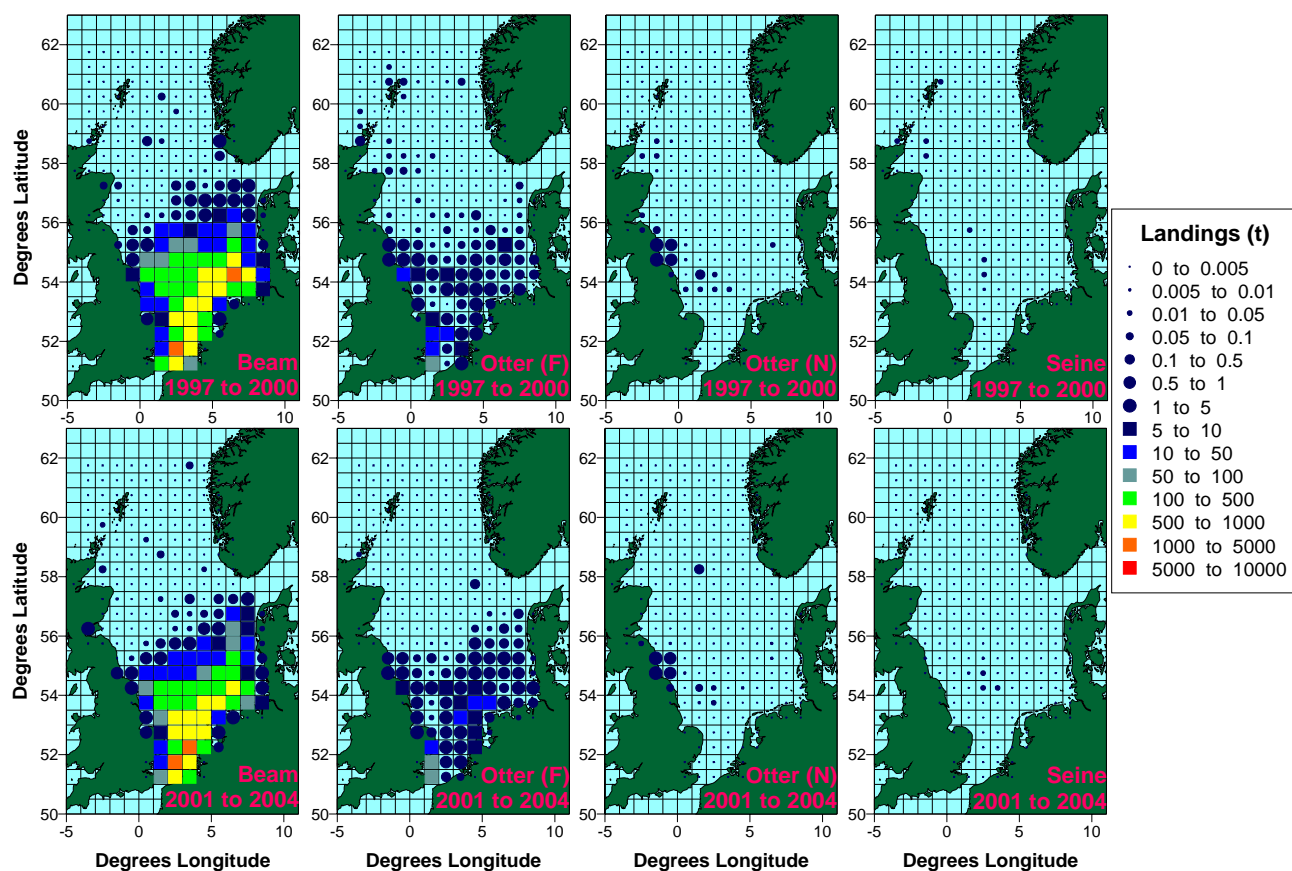


Figure 8.3.2.2.6. Spatial distributions in the average annual landings of sole taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear, in two four year periods, 1997 to 2000 and 2001 to 2004.

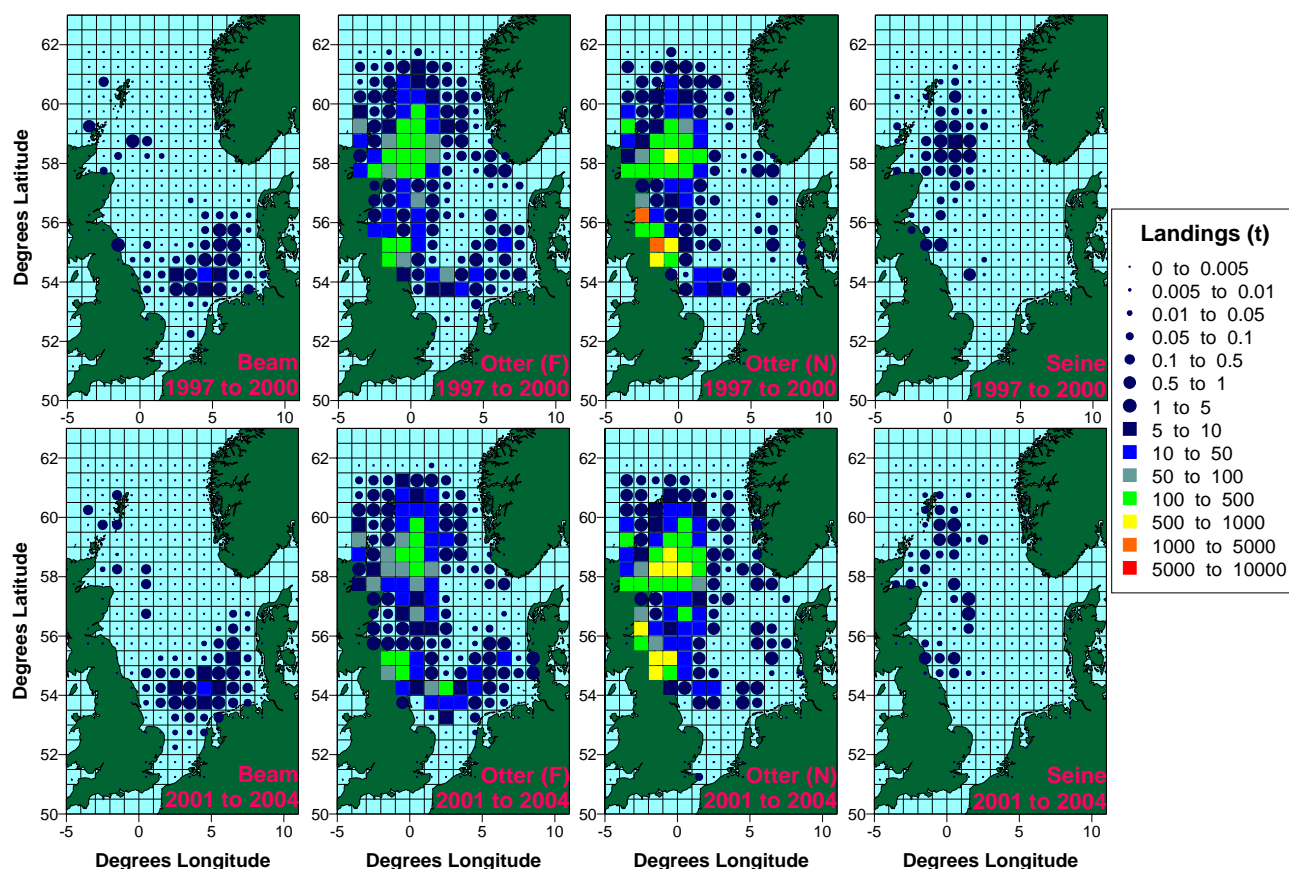


Figure 8.3.2.2.7. Spatial distributions in the average annual landings of *Nephrops* taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear, in two four year periods, 1997 to 2000 and 2001 to 2004.

8.3.2.3. Landings by country (MAFCONS participants)

Because of the temporal trends in total landings illustrated in Figures 8.3.1.1 to 8.3.1.3, and the associated changes over time in the spatial distributions of the landings of each species (Figures 8.3.2.1.1 to 8.3.2.1.7), average annual landings of each species by each of the five countries that contributed to the MAFCONS database were examined for two separate time periods, 1997 to 2000 and 2001 to 2004. Although differences in absolute values were apparent, driven by the temporal variation in total landings (Figure 8.3.1.1), essentially distributions of landings of each species by each of the countries examined were the same in each of the two time periods (Figures 8.3.2.3.1 to 8.3.2.3.7). UK vessels landed cod from all parts of the North Sea, although there was a greater tendency for Scottish vessels to operate preferentially in the north, and English (and Welsh and Northern Irish) vessels to operate in the south. Landings of cod by German and Norwegian vessels were mainly taken from the eastern North Sea, with a tendency for German landings to originate from the southeast and Norwegian landings from the northeast. Dutch cod landings were derived almost entirely from the southern North Sea (Figure 8.3.2.3.1). Similar situations were apparent for haddock, with the exception that almost no haddock were landed by the Netherlands (Figure 8.3.2.3.2), and for whiting, although in this case it was Norway that landed very few fish (Figure 8.3.2.3.3). In the case of saithe, again almost none was landed by the Netherlands. The German and Norwegian landings patterns were almost identical, revealing a strong northeastern tendency.

Scottish and English saithe landings distributions were also similar, showing a strong northern tendency, and were more wide spread than the German and Norwegian distributions (Figure 8.3.2.3.4). Landings distributions of plaice and sole by all countries showed a strong southeastern tendency, particularly in respect of sole (Figure 8.3.2.3.6), although Scottish landings of plaice tended to be more widely distributed than those of the other countries. *Nephrops* landings distributions differed between the five countries. Scottish landings were primarily restricted to the northeastern, English landings to the eastern central, and German and Norwegian landings to the southeastern North Sea (Figure 8.3.2.3.7).

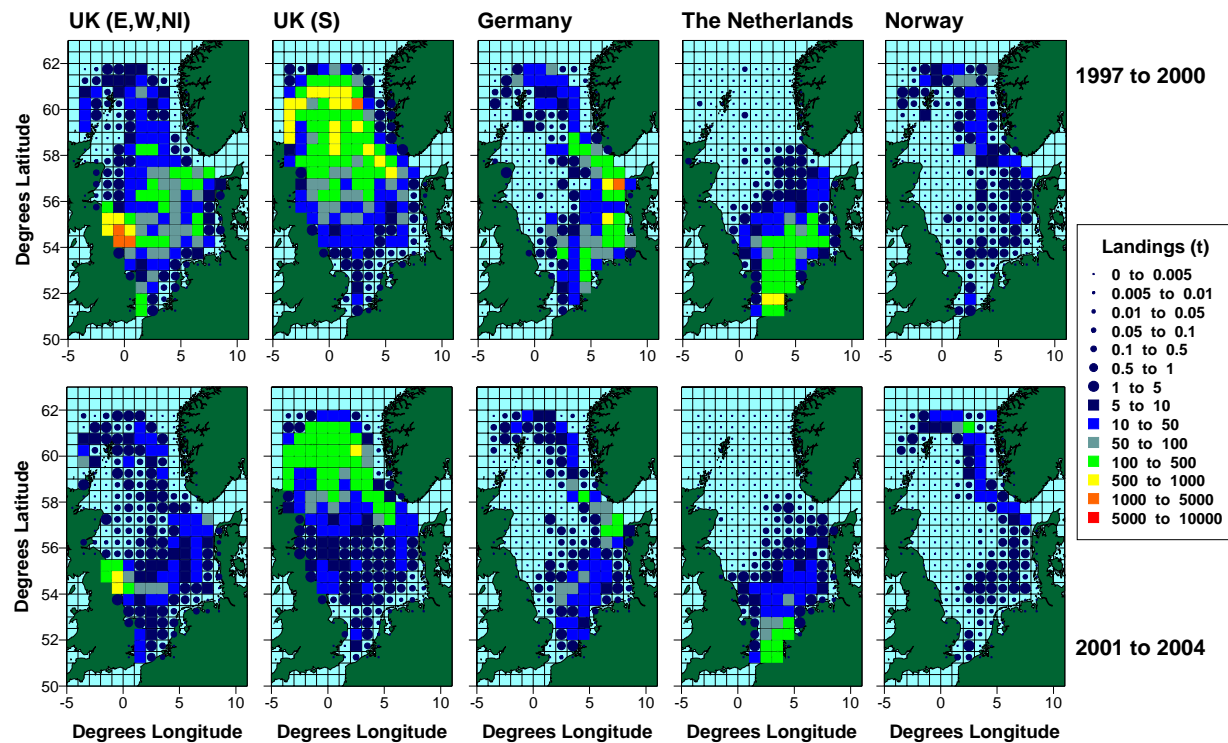


Figure 8.3.2.3.1. Spatial distributions of average annual landings of cod taken by the five countries contributing to the MAFCONS landings database, the UK (England, Wales and Northern Ireland {UK (E,W,NI)}), the UK (Scotland) {UK(S)}, Germany, The Netherlands, and Norway, in two four year periods, 1997 to 2000 and 2001 to 2004.

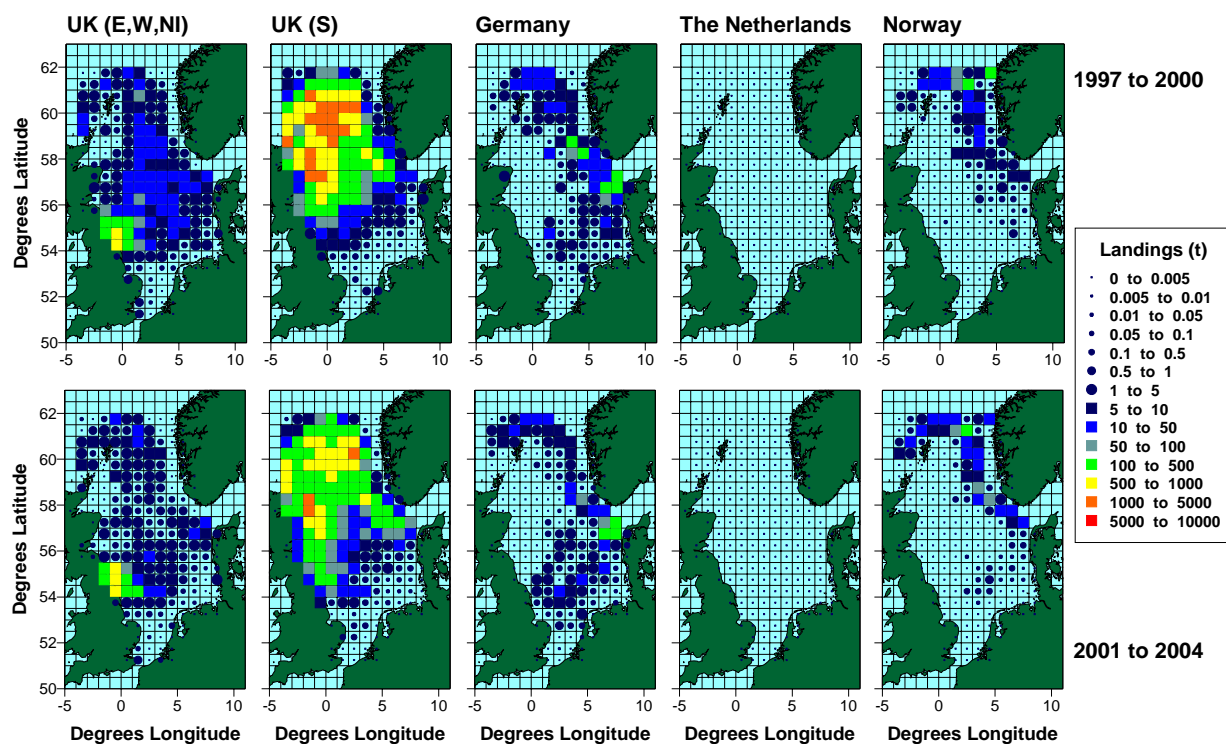


Figure 8.3.2.3.2. Spatial distributions of average annual landings of haddock taken by the five countries contributing to the MAFCONS landings database, the UK (England, Wales and Northern Ireland {UK (E,W,NI)}), the UK (Scotland) {UK(S)}, Germany, The Netherlands, and Norway, in two four year periods, 1997 to 2000 and 2001 to 2004.

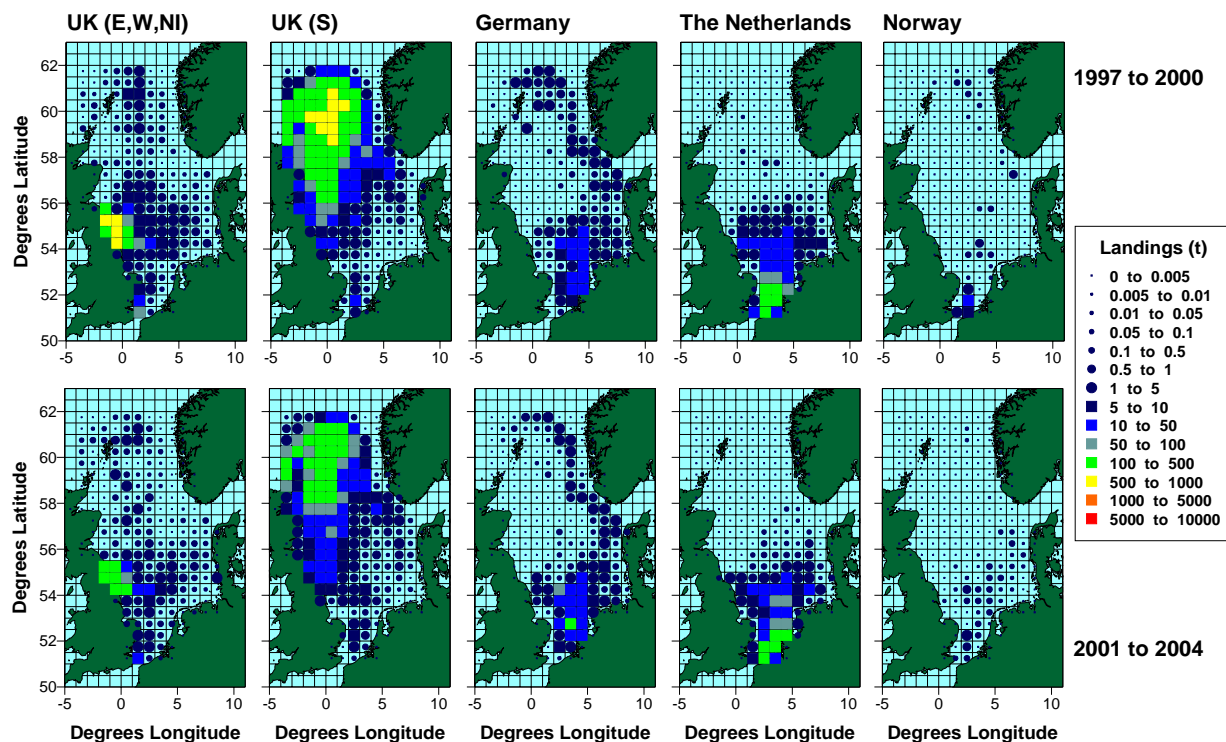


Figure 8.3.2.3.3. Spatial distributions of average annual landings of whiting taken by the five countries contributing to the MAFCONS landings database, the UK (England, Wales and Northern Ireland {UK (E,W,NI)}), the UK (Scotland) {UK(S)}, Germany, The Netherlands, and Norway, in two four year periods, 1997 to 2000 and 2001 to 2004.

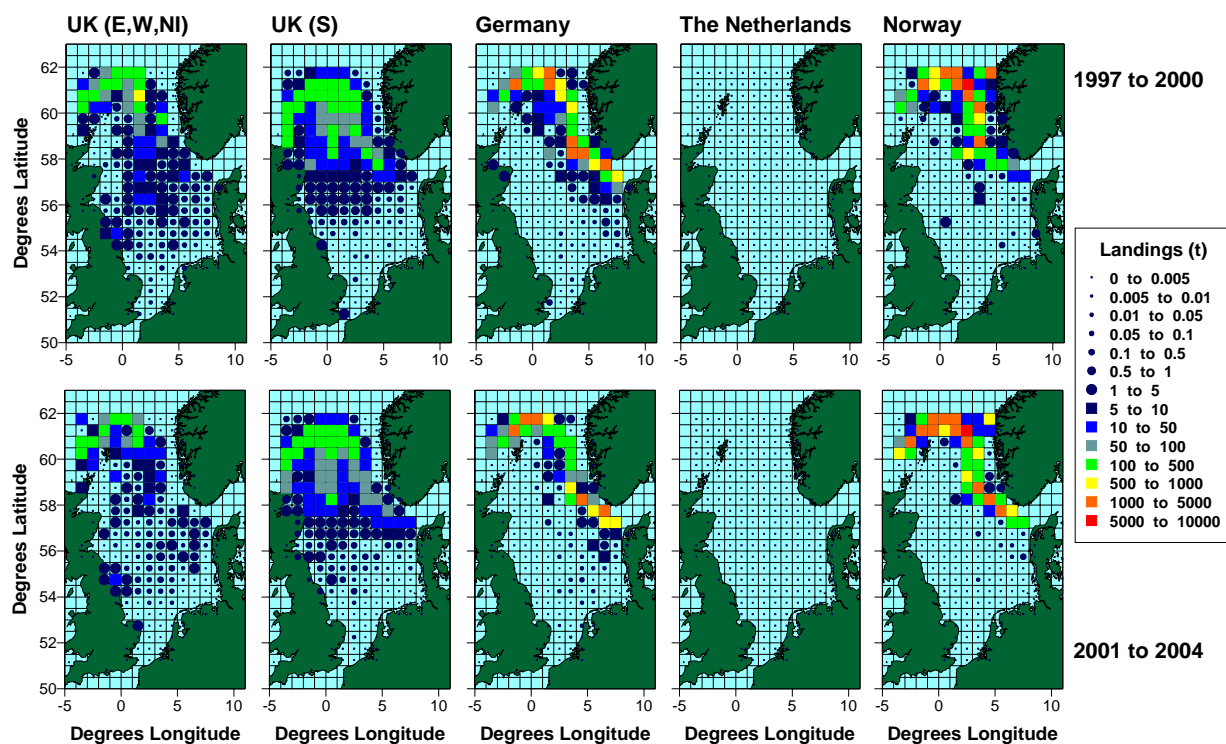


Figure 8.3.2.3.4. Spatial distributions of average annual landings of saithe taken by the five countries contributing to the MAFCONS landings database, the UK (England, Wales and Northern Ireland {UK (E,W,NI)}), the UK (Scotland) {UK(S)}, Germany, The Netherlands, and Norway, in two four year periods, 1997 to 2000 and 2001 to 2004.

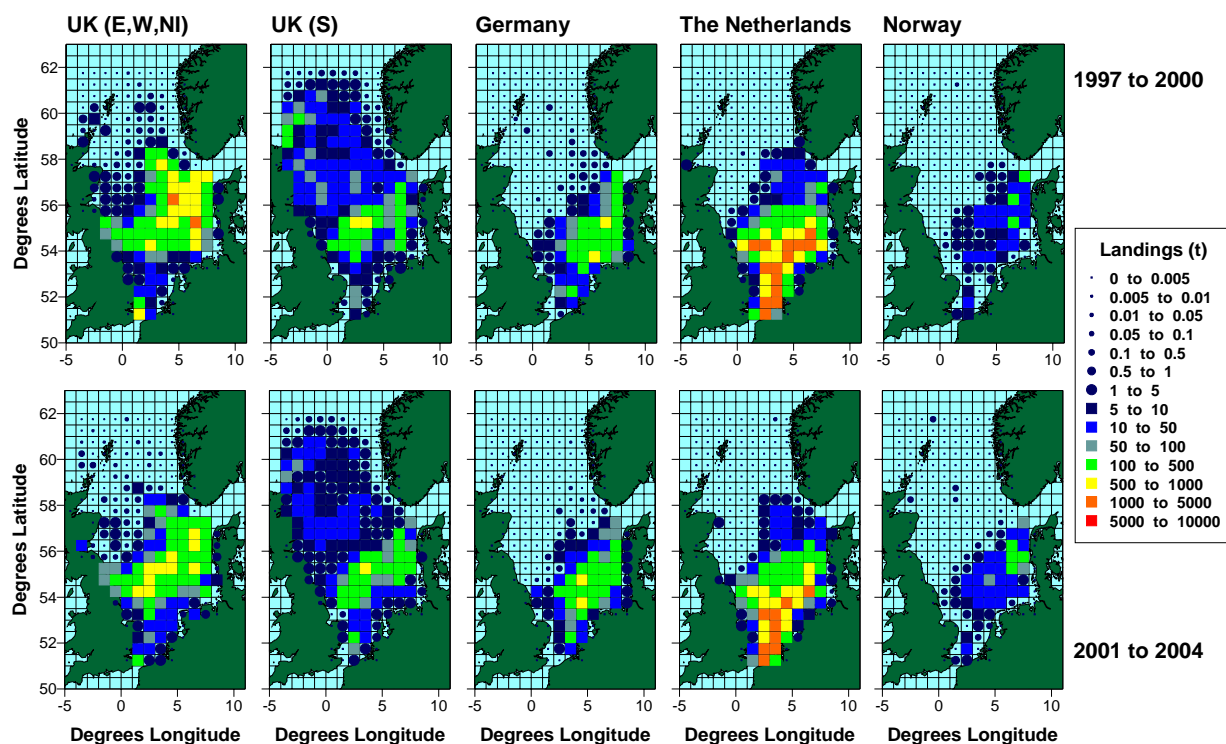


Figure 8.3.2.3.5. Spatial distributions of average annual landings of plaice taken by the five countries contributing to the MAFCONS landings database, the UK (England, Wales and Northern Ireland {UK (E,W,NI)}), the UK (Scotland) {UK(S)}, Germany, The Netherlands, and Norway, in two four year periods, 1997 to 2000 and 2001 to 2004.

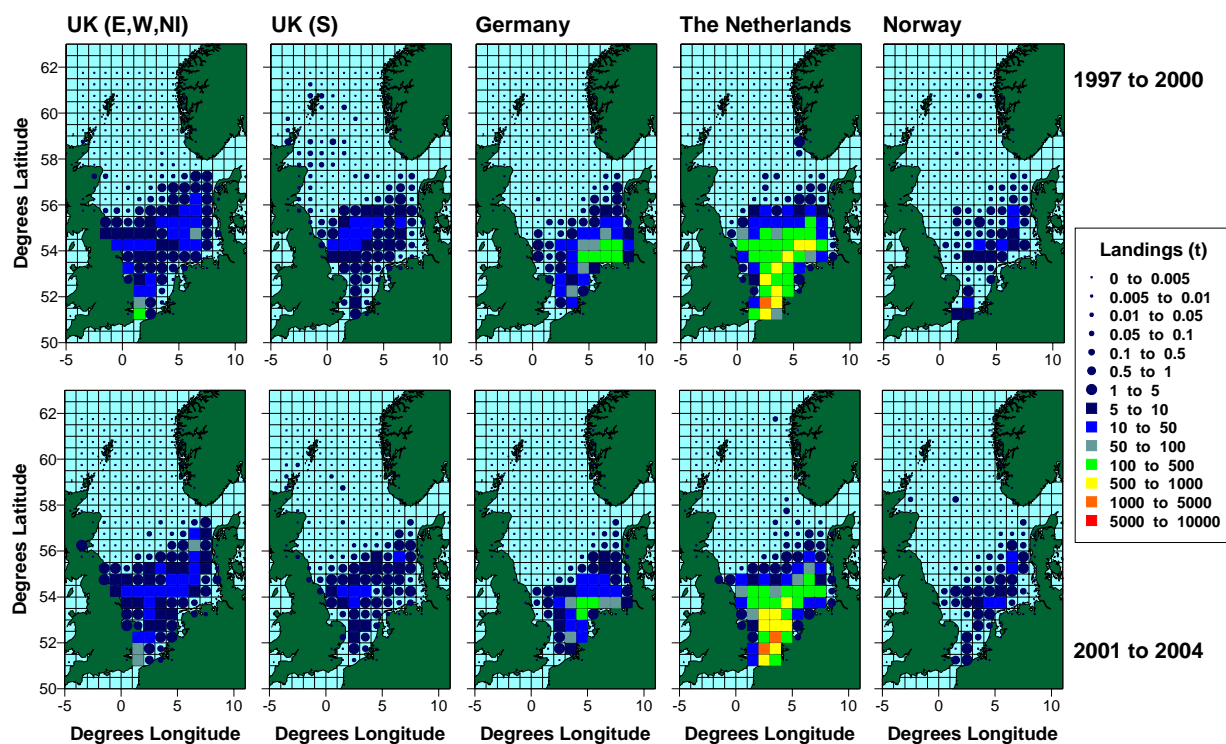


Figure 8.3.2.3.6. Spatial distributions of average annual landings of sole taken by the five countries contributing to the MAFCONS landings database, the UK (England, Wales and Northern Ireland {UK (E,W,Ni)}), the UK (Scotland) {UK(S)}, Germany, The Netherlands, and Norway, in two four year periods, 1997 to 2000 and 2001 to 2004.

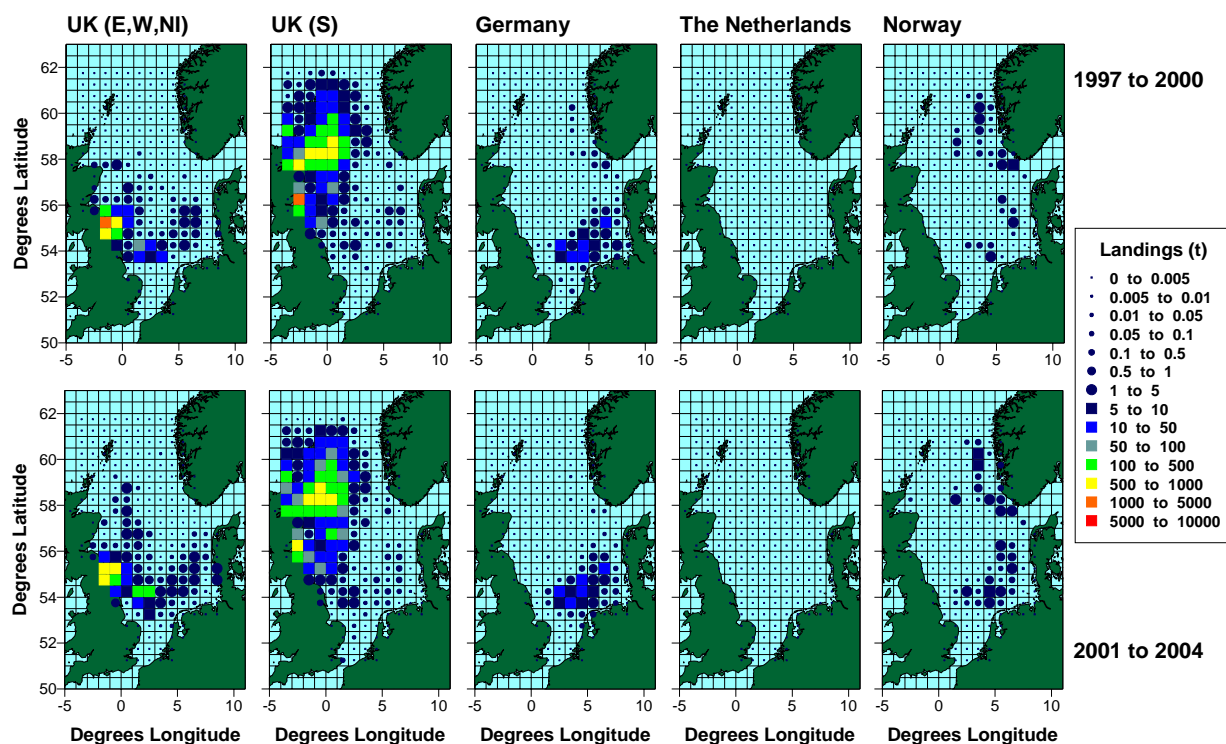


Figure 8.3.2.3.7. Spatial distributions of average annual landings of *Nephrops* taken by the five countries contributing to the MAFCONS landings database, the UK (England, Wales and Northern Ireland {UK (E,W,Ni)}), the UK (Scotland) {UK(S)}, Germany, The Netherlands, and Norway, in two four year periods, 1997 to 2000 and 2001 to 2004.

8.4. Compilation of international effort databases (MAFCONS participants)

Community level changes in both demersal fish and benthic invertebrates have occurred in the North Sea over the last century (For review see Greenstreet et al., 1999; Clark & Frid, 2000; Kröncke & Bergfeld, 2001). It is certain that the disturbance caused by fishing has contributed to these changes (eg Greenstreet & Rogers 2006), and in some cases the mechanistic link between a change (such as a decrease in dominance of species with particular life history characteristics) and a direct effect of fishing (such as size selective mortality) may be clear (see Jennings et al. 1998, 1999a). In order to understand how fishing contributes to community level changes it is vital that we first understand the actual direct ecological disturbance that occurs as a result of fishing. In order to do this we first need to describe the disturbance, and as the first step towards this goal, we need to map the distribution of fishing activity in time and space at scales that are relevant to fish and invertebrate communities.

Landings data quantify the level of fishing impact with respect to each stock, but for other components of the marine ecosystem, these landings data provide little or no information regarding the level of damaging activity taking place. A basic rule equates “catch per unit effort” (*cpue*) to abundance (*N*): $cpue = q N$, where *q* is the catch efficiency of the fishing operation. If a given stock declines in abundance, TACs may be set at lower levels in an attempt to redress the situation. Consider a situation where the size of a stock (*N*) decreases by 66.7% eliciting a reduction in TAC of 50%. Subsequent landings data confirm that the number of individuals taken by the fishery has indeed reduced to 50% of the number caught formerly. However, because *N* is only 33.3% of previous levels, and assuming constant *q*, *cpue* will actually have decreased to 33% of previous levels. To achieve a catch 50% that of earlier levels therefore requires an increase in effort of 150%. Thus although fishing impact on the targeted stock may have come down, mortality of other non-assessed, non-targeted fish and benthic invertebrates, and disturbance to seabed sediment habitats, may have increased by a factor of 1.5, in line with the increase in fishing effort. For components of the marine ecosystem beyond the targeted commercial species therefore, measures of fishing effort provide the most appropriate means of quantifying levels of fishing activity.

Two previous studies have attempted to compile “international effort” databases as hours effort by beam and otter trawlers per ICES rectangle per year. Jennings et al. (1999) assembled data for the period 1990 to 1995, while Callaway et al. (2002) added data for the year 1998. These databases included effort for all vessels fishing in the North Sea and landing into the UK, The Netherlands, Germany, Denmark and Norway. No data were available from France and Belgium. The MAFCONS project has assembled data for UK, Dutch, Norwegian, and German vessels covering the period 1998 to 2004. The major demersal gears used by these fleets were assigned to four main categories of fishing gear; beam trawl, otter trawl directed at fish, otter trawl directed at Nephrops, and Seine gear. Seine gears were not included in the earlier studies. Seine gear has a relatively low impact on benthic organisms, and since these studies were primarily focused on benthic invertebrate communities, Seine gears were not considered to be particularly relevant. However, Seine gears catch considerable amounts of fish in some parts of the North Sea, and so they can potentially affect demersal fish community composition and diversity. Seine gear was therefore also considered in the MAFCONS project.

Data were supplied as hours fishing, by gear, rectangle and year by Norway (by aggregating monthly data for beam and otter trawl gear codes only), Germany (by aggregating monthly data for beam and otter trawl gear codes only), England (and Wales and Northern Ireland for 11 individual gear codes). For these countries little else needed to be done. For Norway and Germany all otter trawl effort data was assumed to be otter trawl directed at fish as the Nephrops quota for both countries was negligible. No Seine gear data were available for either of these two countries, but this gear is certainly not commonly deployed by either country's fleet. Thus for Norway and Germany, all the effort was assigned to either beam trawl or to otter trawl directed at fish. English (and Welsh and Northern Irish) effort was assigned to either of the four main gear categories.

Hours effort data were not available for the Dutch fleet, but days absence from port information was. This was converted to hours fishing assuming a 17 hour "fishing day" (Rijnsdorp et al. 1998; Piet et al. 2000; Piet et al. in press). The total amount of "fishing time" per trip was assigned to the rectangles fished *pro rata* with the reported landings. Thus if landings were reported for two rectangles, and 66% of the total trip's landings were reported from one and 34% from the other, the total estimated fishing time for the trips was likewise assigned to the two rectangles in a 66:34 ratio. Data were only provided for two main gear categories, beam trawl and otter trawl (albeit, each category sub-divided into two vessel-power components). The Netherlands quota for Nephrops amounts to 2.6% of the TAC, thus it is likely that all the Dutch otter trawl effort was in fact directed at fish, but with the fleet given a small quota thereby allowing Nephrops to be landed when taken in the bycatch. Again, seine gears are rarely, if ever, deployed by Dutch fishing vessels.

Hours effort data were available for Scottish fishing vessels, and these have been reported previously (Greenstreet et al., 1999; 2006). However, the completion of this "field" in fishing trip logs is not mandatory. Traditionally the number of hours fishing was estimated on the basis of the number of days fishing per trip and the skipper's verbal report of his daily fishing activity. In the absence of the latter, the number of hours spent fishing per day was estimated based on the inspector's knowledge of each particular fishery. This became more formalised in the early 1980s with the introduction of a logbook system, but notification of the actual number of hours spent fishing was still not compulsory, although many skippers did provide this information. Where these data were missing, fisheries inspectors continued to estimate them following the original procedures (Greenstreet et al., 1999; 2006). Over the last 10 years, the processing of logbook reports for the Scottish fleet has become more centralised and the inspectors responsible do not have the same experience of the local fishing fleets. Thus, when faced with missing values, several different options appear to have been followed. Firstly, some inspectors, rather than entering hours fished based on a "best guess" have stopped entering hours fishing entirely and so, where no record has been made by the fisherman in the logbook, the effort assigned to that particular trip has been interpreted as zero. Alternatively, "guesses" have continued to be made, but with reduced interaction with skippers, and less experience of the fleet, these "guesses" appear to have become increasingly "standardised", ie 10 hours fishing has been entered for each rectangle fished in a trip, regardless of trip duration, or the number of rectangles fished (pers comm., Aileen Shanks and Rob Kynoch, FRS, Marine Laboratory, Aberdeen). Thus, continued use of the Scottish hours-fishing effort data, as done in the earlier studies, would in all likelihood lead to a major underestimation of fishing activity.

To estimate effort data for the Scottish fleet therefore, hours-fishing was modelled following a similar approach to that used to estimate hours-fishing for the Dutch fleet.

8.4.1. Modelling Scottish hours-fishing

Data were extracted from the Scottish FIN database for all voyages by Scottish fishing vessels over the period 1997 to 2004 that recorded catches from at least one ICES statistical rectangle. Table 8.4.1.1 lists the information extracted from FIN used to construct a series of relational databases using PARADOX database software, and indicates the field type. The field “hours fishing” existed in the FIN database and these data were extracted where available. Various anomalies in the data were “cleaned”. For example, power, gross tonnage, and on occasion even vessel length sometimes varied for a particular vessel registration. Where a change was made between voyages, and remained consistent thereafter, these were considered to be real alterations, possibly following a refit for example. More often that not, however, these values changed mid-voyage, or changed and then returned to their original values. In these cases the changes were considered to be anomalies and were altered to be consistent with all other values for that vessel.

Parameter No.	Parameter	Parameter type
1	Voyage identifier code	Relational field
2	Vessel registration	Vessel descriptor
3	Vessel gross tonnage	Vessel descriptor
4	Vessel power	Vessel descriptor
5	Vessel length	Vessel descriptor
6	Date of Departure	Trip descriptor
7	Date of Return	Trip descriptor
8	Date of Landing	Trip descriptor
9	Days absent from port	Trip descriptor
10	Port of Departure	Trip descriptor
11	Port of Return	Trip descriptor
12	Port of Landing	Trip descriptor
13	FRS Gear Code	Gear descriptor
14	Minimum mesh size	Gear descriptor
15	ICES rectangle	Landings data
16	<i>Hours fishing in rectangle</i>	<i>Effort data</i>
17	Landed weight of cod caught	Landings data
18	Landed weight of haddock caught	Landings data
19	Landed weight of whiting caught	Landings data
20	Landed weight of saithe caught	Landings data
21	Landed weight of plaice caught	Landings data
22	Landed weight of sole caught	Landings data
23	Landed weight of anglerfish caught	Landings data
24	Landed weight of <i>Nephrops</i> caught	Landings data
25	Total weight demersal fish caught	Landings data
26	Total weight shellfish caught	Landings data

Table 8.4.1.1. List of parameter values obtained from FIN database

Anomalies were also apparent in the fishing gear codes recorded in the FIN database. For some voyages two separate gears were recorded for a single rectangle, thus for example a *Nephrops* trawl code and a gill net code, for example, would both be recorded, and in all other rectangles fished

during the voyage, the *Nephrops* trawl would be indicated. Examination of the landing data in these instances indicated no obvious difference in catch composition between the two very different gears. Under these circumstances the gill net code was deleted from the PARADOX data base and the catch assigned to gill net added to the catch associated with the *Nephrops* trawl code. An alternative, but less obvious anomaly was the recording of both *Nephrops* and fish otter trawl codes in a single rectangle during a voyage, by vessels that predominantly used one or other of these codes. In these cases, one of the codes was changed to comply with the codes predominantly used for the remainder of the voyage and the landings were reassigned accordingly. Again examination of the landings data confirmed that the decisions taken were sensible. In making these changes we aimed to have only one gear used in each rectangle visited within a voyage by each vessel. In total, gear codes were changed in only 0.2% of the total number of records in the resulting PARADOX database. Any influence on our results is likely therefore to be minimal, but these changes greatly simplified our calculations. Table 8.4.1.2 lists the FRS demersal fishing gear codes for which data were available.

FRS Gear Code	Gear Description	Main Gear Category
MTR	Heavy otter trawl (fish)	Otter trawl for fish
LTR	Light otter trawl (fish)	Otter trawl for fish
MTD	Multiple otter trawl (fish)	Otter trawl for fish
ITR	Industrial otter trawl (fish)	Otter trawl for fish
PTD	Demersal pair trawl (fish)	Otter trawl for fish
SEN	Seine gear	Seine gear for fish
PSN	Pair seine gear	Seine gear for fish
NTR	Single <i>Nephrops</i> trawl	Otter trawl for <i>Nephrops</i>
MTN	Multiple <i>Nephrops</i> trawl	Otter trawl for <i>Nephrops</i>
BTR	Beam trawl	beam trawl for fish
QTR	Queen scallop trawl	Otter trawl for invertebrates
STR	Shrimp trawl	Otter trawl for invertebrates

Table 8.4.1.2. Description of 13 demersal gear codes for which landings data were available in the FIN database

Figure 8.4.1.1A shows the frequency distribution for days absent from port for all trips fishing in at least one North Sea ICES statistical rectangle. 50% of voyages lasted only one or two days and for a further 30% of voyages, the vessel was absent from port for between three and seven days. “Days absent from port” (DA) in the database included both the day of departure and the day of return. This was deemed necessary because many voyages left and returned to port within the same day, thus the database gave these voyages a “days absent” score of one day. However, consider an extreme example of a voyage departing at 2300h on one day and returning at 0100h the following day. Such a trip would be given a “days absent” score of two days, for a voyage actually lasting for only two hours. Ultimately our intention was to model hours fishing, so a simple scaling rule was used to convert “days absent from port” to “voyage working hours” (T_{WORK}) (Table 8.4.1.3), where “working” included both steaming time between port and fishing grounds (T_{STEAM}) and fishing time (T_{FISH}),

$$T_{WORK} = T_{STEAM} + T_{FISH} \quad 8.4.1.1.$$

In essence, for trips of 6 days absence or more, this rule assumes that the vessel departed and returned at the same time of day, so that only one full day was considered available for working out of the two days, the day of departure and the day of return. For shorter duration trips a sliding scale was assumed.

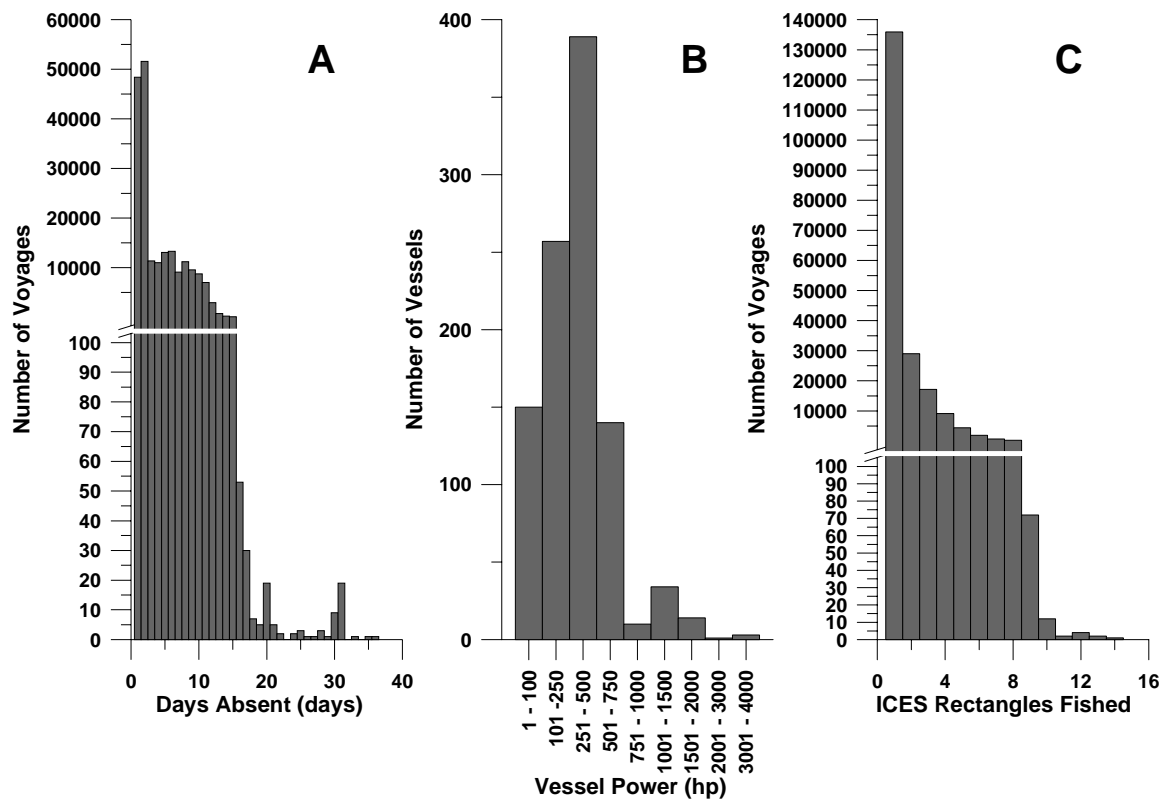


Figure 8.4.1.1. Frequency distributions of days absent from port (A), numbers of Scottish fishing vessels assigned to various power categories (B), and ICES rectangles visited per voyage (C) for 198,476 voyages by Scottish vessels that fished in at least one ICES statistical rectangle in the North Sea.

“Days absent”	“Hours working” formulation	“Hours working” (T_{WORK})
$DA = 1$	Round ($da \times 24 \times 0.875$)	21
$DA = 2$	Round ($da \times 24 \times 0.875$)	42
$DA = 3$	Round ($da \times 24 \times 0.850$)	61
$DA = 4$	Round ($da \times 24 \times 0.825$)	79
$DA = 5$	Round ($da \times 24 \times 0.815$)	98
$DA = 6, 7, \dots$	$(da - 1) \times 24$	120, 144, ...

Table 8.4.1.3. Scaling rule used to convert days absent from port to voyage working time.

Voyages visiting ICES rectangles far from port require a greater proportion of time spent steaming to the fishing grounds. Therefore, simply scaling “fishing effort” to “days absent from port”, or “hours working” was not an option since this would introduce a spatial bias to the resultant distributions of fishing activity. Fishing activity in far offshore ICES rectangles would be systematically over-estimated compared with ICES rectangles closer to port. Minimum steaming distances (D_{MIN}) were therefore calculated for each voyage as:

$$D_{MIN} = D_{OUT} + D_{IN} \quad 8.4.1.2.$$

where D_{OUT} is the maximum outward distance, the distance between the port of departure and the ICES rectangle furthest away from this port, and D_{IN} is the maximum inward distance, the distance between the port of return and the ICES rectangle furthest away from this port.

A total of 198,476 voyages by Scottish fishing vessels landed fish taken from at least one ICES statistical rectangle in the North Sea. Although ICES rectangles in the North Sea were therefore the most frequently visited over the course of these voyages, numerous rectangles outside the North Sea were also visited a substantial number of times (Figure 8.4.1.2). In total, 394 ICES rectangles were visited. Furthermore, the vessels making these voyages departed and returned from a total of 146 different ports. These were situated primarily in Scotland (108 ports), but also in England, Wales and Ireland (20 ports), around the North Sea coast of continental Europe (14 ports), Spain (1 port), the Faeroes (2 ports) and Iceland (1 port). A number of rules were therefore applied in calculating both D_{IN} and D_{OUT} (Table 8.4.1.4).

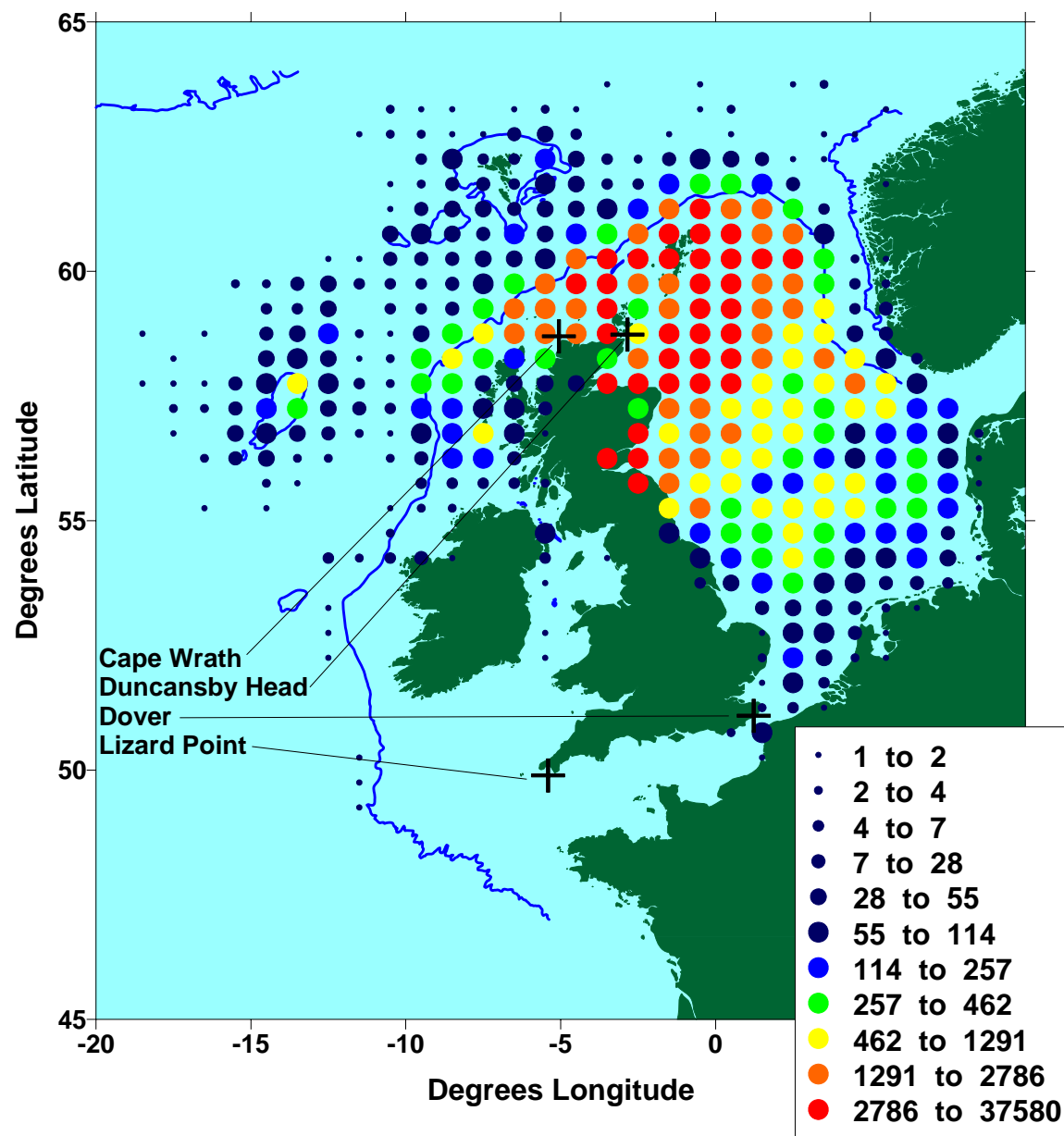


Figure 8.4.1.2. Classed post map indicating the frequency that each of the 349 ICES rectangles was visited during 198,476 voyages by Scottish vessels that fished in at least one ICES statistical rectangle. Waypoints used for routing vessels between port and fishing grounds (see Table 4) are indicated.

Location of fished rectangle	Location of departure/return port	Routing rule
North Sea	North Sea coast	Shortest straight line
	West of Britain south of 54°N and Spanish port	Shortest distance via waypoints at Lizard Point (50° 04.01'N 005° 44.81'W) and Dover (51° 06.47'N 001° 20.61'E)
	West of Britain north of 54°N and Icelandic and Faeroese ports	Shortest distance via waypoints at Cape Wrath (58° 37.54'N 004° 59.99'W) and Duncansby Head (58° 38.65'N 003° 01.58'W)
North of 61°N and east of 004°W	North Sea coast and Icelandic ports	Shortest straight line
	West of Britain and Spanish port	Shortest distance via waypoint at Cape Wrath
South of 51°N and east of 000°E	North Sea coast	Shortest straight line
	West of Britain and Spanish and Icelandic ports	Shortest distance via waypoint at Lizard Point
North of 58° 30'N and west of 004°W	North Sea coast	Shortest distance via waypoint at Duncansby head
	West of Britain and Spanish and Icelandic ports	Shortest direct route
Between 52°N and 58° 30'N and west of 003°W	North Sea coast	Shortest distance via waypoints at Duncansby Head and Cape Wrath
	West of Britain and Spanish and Icelandic ports	Shortest direct route
South of 52°N and west of 003°W	North Sea coast	Shortest distance via waypoint at Dover
	West of Britain and Spanish and Icelandic ports	Shortest direct route

Table 8.4.1.4. Routing rules used to determine maximum outward distance between port of departure and furthest away ICES rectangle fished (D_{OUT}) and maximum inward distance between port of return and furthest away ICES rectangle fished (D_{IN}).

These fishing voyages were made by 989 different vessels with a wide range in horsepower from 9hp to 3966hp (Figure 8.4.1.1B). Five power bands were defined and the cruising speeds (S_{STEAM})

assumed for each power band are indicated in Table 5. The minimum time that vessels must spend steaming between port and fishing grounds in each voyage (T_{STEAM}) was then determined by:

$$T_{STEAM} = D_{MIN} / S_{STEAM} \quad 8.4.1.3.$$

By rearranging and combining these equations, the maximum potential time available for fishing on each voyage was calculated by:

$$T_{FISH} = T_{WORK} - \left(\frac{D_{IN} + D_{OUT}}{S_{STEAM}} \right) \quad 8.4.1.4.$$

In the case of 211 voyages (0.11% of the total number of voyages), all of short duration, four days absent from port or less, and mostly involving small vessels <17m in length, negative fishing time was indicated following application of equation 1 to their data ($T_{STEAM} > T_{WORK}$). All of these voyages landed fish, so fishing had clearly taken place. Some error in data recording had obviously occurred in respect of these voyages, leading to the presence of erroneous data in the database. For these voyages therefore, the vessel was arbitrarily considered to have fished for one hour of each day that it was absent from port.

Vessel power band (hp)	Cruising speed (knots)	Cruising speed (km h ⁻¹)
1-250	7	13
251-500	8	15
501-1000	9	17
1001-2000	10	19
2001-4000	12	22

Table 8.4.1.5. Cruising speeds (V_{STEAM}) assumed for vessels assigned to five power bands.

Figure 8.4.1.1C indicates the frequency distribution for the number of ICES statistical rectangles fished in each voyage. Two methods of apportioning the maximum potential fishing time (T_{FISH}) in each voyage to the ICES rectangles fished were examined. First we assumed that T_{FISH} was split evenly between the rectangles fished, allowing catch per unit effort ($cpue$, where effort is equivalent to time fishing) to vary between rectangles (Time constant, $cpue$ variable method). Second, we assumed that T_{FISH} across the rectangles visited was distributed in direct proportion to the total landings recorded from each rectangle, thus effectively maintaining constant $cpue$ across the rectangles visited, but forcing time to vary between rectangles (time variable, $cpue$ constant method). Both methods were used to assign T_{FISH} for each voyage to the rectangles visited. Then, summing across all voyages where a reported hours fishing was recoded in the database for each rectangle visited, total T_{FISH} by each gear category in each rectangle in each year was determined, along with total reports hours fishing. A regression analysis was then performed with total T_{FISH} as the independent variable and total reported hours fishing as the dependent variable (Table 8.4.1.6). All 24 regression analyses were highly significant, with $p < 0.001$ in every case. Both methods of assigning voyage T_{FISH} to the rectangles fished in each voyage provided a estimate of fishing time that, at the ICES rectangle scale, successfully captured the signal of annual variation in reported hours fishing by each gear. However, Method 1, allowing $cpue$ to vary and distributing T_{FISH} evenly between the rectangles visited, appeared to perform marginally better; correlation coefficients were either the same for both methods (one gear) or higher when method 1 was used (11 gears) (Table 8.4.1.6).

FRS gear code	1. Time constant / $cpue$ variable	2. Time variable / $cpue$ constant
---------------	------------------------------------	------------------------------------

	b	r^2	b	r^2
LTR	0.196	0.917	0.183	0.880
MTR	0.215	0.901	0.186	0.824
MTD	0.165	0.871	0.151	0.825
PTD	0.195	0.956	0.186	0.924
ITR	0.378	0.946	0.369	0.934
SEN	0.273	0.909	0.260	0.864
PSN	0.234	0.949	0.213	0.904
BTR	0.325	0.805	0.290	0.754
NTR	0.274	0.970	0.274	0.969
MTN	0.181	0.974	0.175	0.960
STR	0.139	0.935	0.057	0.846
QTR	0.166	0.943	0.166	0.943

Table 8.4.1.6. Results of regression analysis examining the relationships between total T_{FISH} and total reported hours fishing (by rectangle and year) for each FRS fishing gear code after using two methods (Time constant / $cpue$ variable and Time variable / $cpue$ constant) to assign T_{FISH} for each voyage to the ICES rectangles visited.

Despite the apparent success of using T_{FISH} to predict reported hours fishing in each rectangle in each year, Table 8.4.1.6 nevertheless indicates a potential problem. Published studies of fishing activity suggest that fishermen spend a high proportion of their time on fishing grounds with their gear in the water fishing (eg Piet et al 2000; in press), whereas the regression coefficients in Table 8.4.1.6 would tend to suggest that only around 15% to 40% of the time available for fishing on each voyage was spent with the gear in the water. T_{FISH} appears to provide a good relative index of fishing activity, but seems markedly to underestimate absolute reported fishing activity. The low b values derived from the regression analysis suggest that using reported hours-fishing, where available in the database, to parameterise a model converting T_{FISH} to an index of absolute fishing activity would certainly not be reliable. To understand why this should be the case we examined the data from all voyages where only one rectangle was fished, ie with the simplest trip structure possible; port to fishing-ground to port. For such voyages, a clear relationship between days absent from port and the hours reported fishing should have been observed, but this was not the case (Figure 8.4.1.3). For voyages of between 1 and 13 days duration, the mean hours reported fishing was remarkably constant at between 8 and 12 hours. This was strongly indicative of a constant value (eg 10h) being reported/recorded for a large proportion of the voyages, with this value bearing little relation to the actual amount of fishing activity that had occurred during these trips. It is simply beyond belief that fishing voyages of nearly two weeks duration should spend as little as 10 to 12 hours actually fishing. It is no surprise therefore that estimates of total fishing activity per ICES rectangle, based on reported hours fishing, when related to T_{FISH} produce regression coefficients as low as those shown in Table 8.4.1.6.

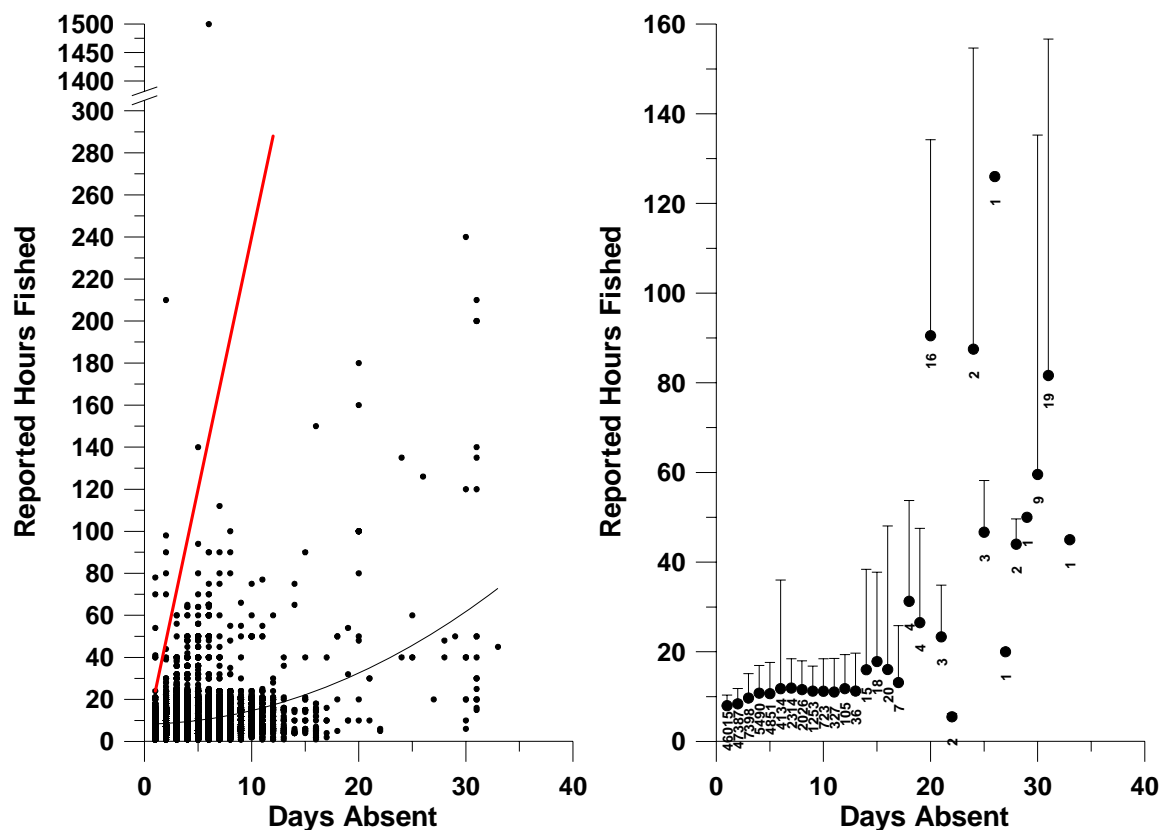


Figure 8.4.1.3. Two plots illustrating the relationship between reported hours fishing and the days absent from port for fishing voyages fishing only one ICES rectangle and where an values for hours fished is provided in the database. Left panel shows the data for each individual trip, with the red line indicating the absolute maximum possible (ie 24 hours per day absent from port). Fitted curve shows a polynomial relationship ($r^2=0.06$). Right panel shows mean (+1SD) Reported hours fishing for voyages of given days absent from port.

Figure 8.4.1.3 also reveals other major anomalies in the reported hours fishing data. For example, one voyage of 6 days absent from port has 1500 hours fishing reported. This is obviously an error, but several other trips of one or two day's duration also have more hours reported fishing than they were actually away from port for! Voyages where the vessel was absent from port for 14 days or longer tended to have more hours reported fishing. Nevertheless, the 16 voyages absent from port for 20 days still only reported an average of 90.5 hours fishing; less than 20% of the total voyage duration. More disturbing still is the cluster of voyages of 30 ($n=9$) and 31 ($n=19$) days absent from port. These voyages only reported means of 60 and 82 hours fishing respectively; 8% and 11% of the total time away from port. One might have expected that the vessels involved in such long voyages would have been among the largest in the Scottish fishing fleet, yet examination of the data revealed quite the opposite. These 28 voyages involved only four vessels, all less than 12m in length and less than 180hp, with each voyage departing and returning to the same port. All voyage start dates were the first day of the month and the return/landings dates were the last day of the month. Rather than being voyage records, it would appear that these records in the FIN database referred to monthly landings records by small vessels almost certainly operating on a day by day basis. Examination of the entire database revealed that 84% of the 68 voyages absent from port for 20 days or longer involved vessels <12m in length. It is inconceivable that vessels of this size can make voyages of such duration. All of these records were therefore also likely to be “periodic” landings records rather than records relating to single voyages.

Since the modelling approach adopted in this study can not derive estimates of hours-fishing from such monthly, or “periodic”, landings records, it was necessary to determine the extent to which such records pervaded the database. In a previous paper (Greenstreet et al 1999), only effort for vessels of $\geq 17\text{m}$ in length was reported because at the time different log-book regulations applied to smaller vessels. More recently the EC has stipulated that only vessels $< 12\text{m}$ in length were exempt from the regulations requiring fishing activity to be reported on a voyage by voyage basis. In examining this issue therefore, we considered vessels $< 12\text{m}$ and vessels $\geq 12\text{m}$ separately. A total of 989 different Scottish vessels were involved in fishing in at least one North Sea ICES rectangle during the period 1997 to 2004. Of these vessels, 234 (24%) were $< 12\text{m}$ in length (Figure 8.4.1.4A) and these vessels were responsible for 57,708 (29%) of the fishing voyage records. Trip duration for the larger vessels was relatively evenly distributed, with few voyages lasting more than 16 days (Figure 8.4.1.4B). These longer duration voyages tended to involve long-distance trips, often involving ports in Spain or North Sea European ports and often including visits to fishing grounds west of Scotland. There is no evidence to suggest that these longer duration records, or indeed any of the records, for the larger vessels represented anything other than single voyages. In contrast, data for the smaller vessels were strongly skewed with 98% of voyages absent from port for either one or two days, a realistic duration for vessels of this size (Figure 8.4.1.4C). So, for many of these vessels single voyage data would in fact appear to have been reported. However, the “tail” of apparently longer-duration voyages was unlikely to consist of single-voyage records and almost certainly involved “periodic” landings records.

Database records for vessels $< 12\text{m}$ in length reporting one or two days absent from port accounted for a total of 85,767 days when fishing may have occurred. The 1251 records for vessels $< 12\text{m}$ in length purporting to reflect voyages of 3 or more days absent from port account for a maximum of 7209 days in which fishing could have occurred. If these records do relate predominantly to “periodic” reports rather than single voyages, then T_{FISH} will over-estimate the actual amount time potentially available for fishing. Only single return trips between port and fishing grounds will have been assumed for these records, no time will have been allowed for additional return trips, so T_{STEAM} will be under-estimated. It is also extremely unlikely that the vessels would have been at sea for the maximum possible time during these periods (eg a month) for which they reported landings so T_{WORK} will also have been over-estimated. If T_{FISH} was in fact close to zero for these periodic records, ie the most extreme over-estimate possible and a most unlikely scenario given that fish were landed, then the maximum possible error could only amount to approximately 8% ($7209 / (85767 + 7209)$). Assuming that T_{FISH} was over-estimated by a factor of two, error in estimation of T_{FISH} resulting from the presence of these “periodic” landings records was unlikely to exceed 4%. Since any procedure to correct this over-estimate of T_{FISH} would have had little impact on the eventual results, this source of error was ignored.

Examination of the total amount of fish and shellfish landed over the period 1997 to 2004 by vessels $< 12\text{m}$ and $\geq 12\text{m}$ in length revealed a strong dichotomy between the two vessel length classes (Figure 8.4.1.4D). Vessels $\geq 12\text{m}$ in length accounted for 99.9% of all the demersal fish landed. However smaller fishing vessels, $< 12\text{m}$ in length, accounted for a larger fraction (10%) of the shellfish landed. To a considerable degree therefore, this suggests that the impact of fishing vessels $< 12\text{m}$ in length on the marine habitats and communities of the North Sea must be relatively minor. However, the considerable difference in *cpue* between these two categories of vessels suggests that,

if activity by the smaller vessels is to be modelled at all, then the activity of vessels $<12\text{m}$ and $\geq 12\text{m}$ in length should be modelled separately.

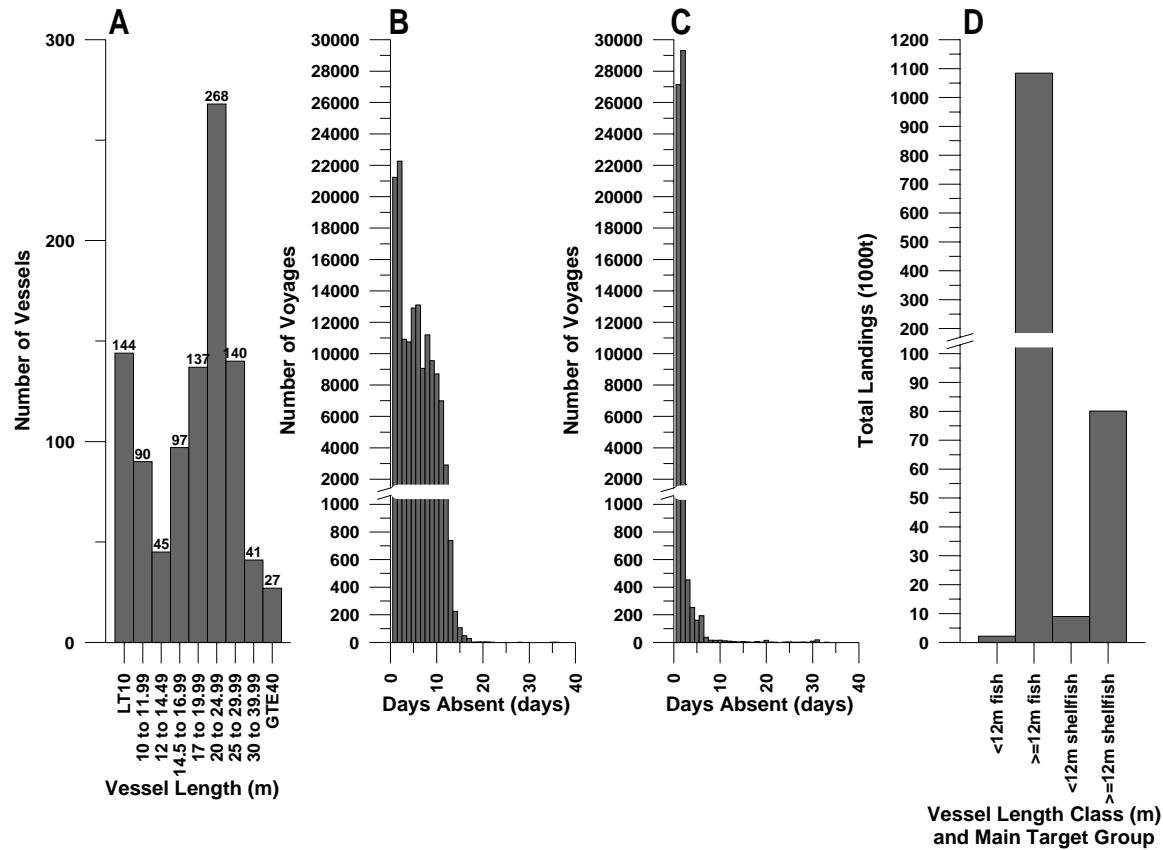


Figure 8.4.1.4. A: Frequency distribution showing the number of different Scottish fishing vessels belonging to various length categories. B: Frequency distribution showing the number of database records (“voyages”) reporting a given days absent from port by vessels $\geq 12\text{m}$ in length. C: Frequency distribution showing the number of database records (“voyages”) reporting a given days absent from port by vessels $<12\text{m}$ in length. D: Histogram indicating total landings of demersal fish and shellfish by vessels $<12\text{m}$ and $\geq 12\text{m}$ in length.

Figure 8.4.1.3 and the regression coefficients presented in Table 8.4.1.6 clearly demonstrate that it was not possible to use the hours-fishing data recorded in the FIN database to estimate the proportion of the available potential fishing time (T_{FISH}) that was actually spent fishing during each voyage. Instead data recorded by scientists participating in the 2004 Scottish Discard Observer Scheme were used to obtain estimates of these parameter values for different types of fishing gear. For each voyage carrying an observer, T_{WORK} was determined from the recorded days absent from port, which again included both the dates of departure and return, following the procedure indicated in Table 8.4.1.3. The distance steamed in each observed voyage was determined and T_{STEAM} estimated assuming a cruising speed 15km h^{-1} . The potential fishing time available in each observed voyage (T_{FISH}) was then determined using equation 8.4.1.1. The proportion of available fishing time with gear actually fishing (P_{FISH}) was then determined from the fishing effort actually reported in each of the observer trips. The 2004 observer data provided parameter values for five gears (Table 8.4.1.7). From this information, parameter values for the remaining seven gear codes were derived following the procedures indicated (Table 8.4.1.7). In the absence of other information, P_{FISH} for each gear was assumed to be the same for both vessel length categories. The different $cpue$ between

vessel categories was more likely to reflect variation in the size of gear used rather than difference in the proportion of available time actually fishing.

Gear Code	Proportion of potential fishing time actually fishing (P_{FISH})	Information source for parameter estimate
MTR	0.88±0.06	Scottish Discards Observer data
LTR	0.65±0.08	Scottish Discards Observer data
PTD	0.37±0.04	Scottish Discards Observer data
NTR	0.58±0.04	Scottish Discards Observer data
SEN	0.27±0.03	Scottish Discards Observer data
MTD	0.77	Average of LTR and MTR
MTN	0.68	Same ratio twin to single gear as for gears directed at fish, ie (MTD/LTR) * NTR
ITR	0.65	Same as LTR
PSN	0.20	Marginally higher than 0.5 * SEN
QTR	0.61	Slightly less than LTR reflecting longer sorting time
STR	0.61	Slightly less than LTR reflecting longer sorting time
BTR	0.70	Piet et al (2000)

Table 8.4.1.7. Proportion of potential available fishing time per fishing trip actually spent fishing by Scottish fishing vessels using different demersal fishing gears. Table 8.4.1.2 gives gear description for each code. The source or derivation of the parameter values used for each gear is indicated.

Actual hours-fishing (H_{FISH}) in each voyage was modelled as

$$H_{FISH} = P_{FISH} * T_{FISH} \quad 8.4.1.5.$$

where T_{FISH} was calculated following equation 4. In each voyage, T_{FISH} (and hence, H_{FISH}) was divided evenly between the ICES rectangles with landings reported. Model performance was assessed for each fishing gear by comparing modelled actual hours-fishing with reported hours-fishing in each ICES rectangle in 1997 and 1998, the last two years when the reported hours-fishing were considered to be reasonably reliable (Greenstreet et al 2006). In this study, hours-fishing were analysed only for the vessels covered by the Common Fisheries Policy regulations, ie greater than 12m in length, so initially we examined modelled fishing effort only for this vessel length category. With the exception of beam trawl (BTR), modelled effort (hours-fishing per ICES rectangle per year) provided a reliable indication of reported fishing effort (for all other gears, $r^2 > 0.85$, $p < 0.0001$) (Figure 8.4.1.5). In calculating the regression coefficients, all zero-zero data pairs were excluded. Whilst these were valid data (zero hours-fishing predicted by the model and zero hours-fishing reported), such data pairs were common so their inclusion would have inflated the r^2 values. Excluding these data pairs therefore rendered the test more conservative. Despite the exclusion of zero-zero data pairs from the regression analysis, for all but the beam trawl intercept values were close to zero, so the model did well at predicting low and zero effort rectangles (Figure 8.4.1.5). For three gears, light otter trawl (LTR), multiple otter trawl for fish (MTD), and multiple otter trawl for *Nephrops* (MTM), the regression coefficients were close to one indicating almost exact agreement between modelled hours-fishing and reported hours-fishing. For other gears, the match between modelled and reported effort was less close, but in most instances differences were easily explained.

Regression coefficients greater than one indicate that the model under-estimates reported hours-fishing, while coefficients less than one imply that the model is over-estimating actual fishing effort.

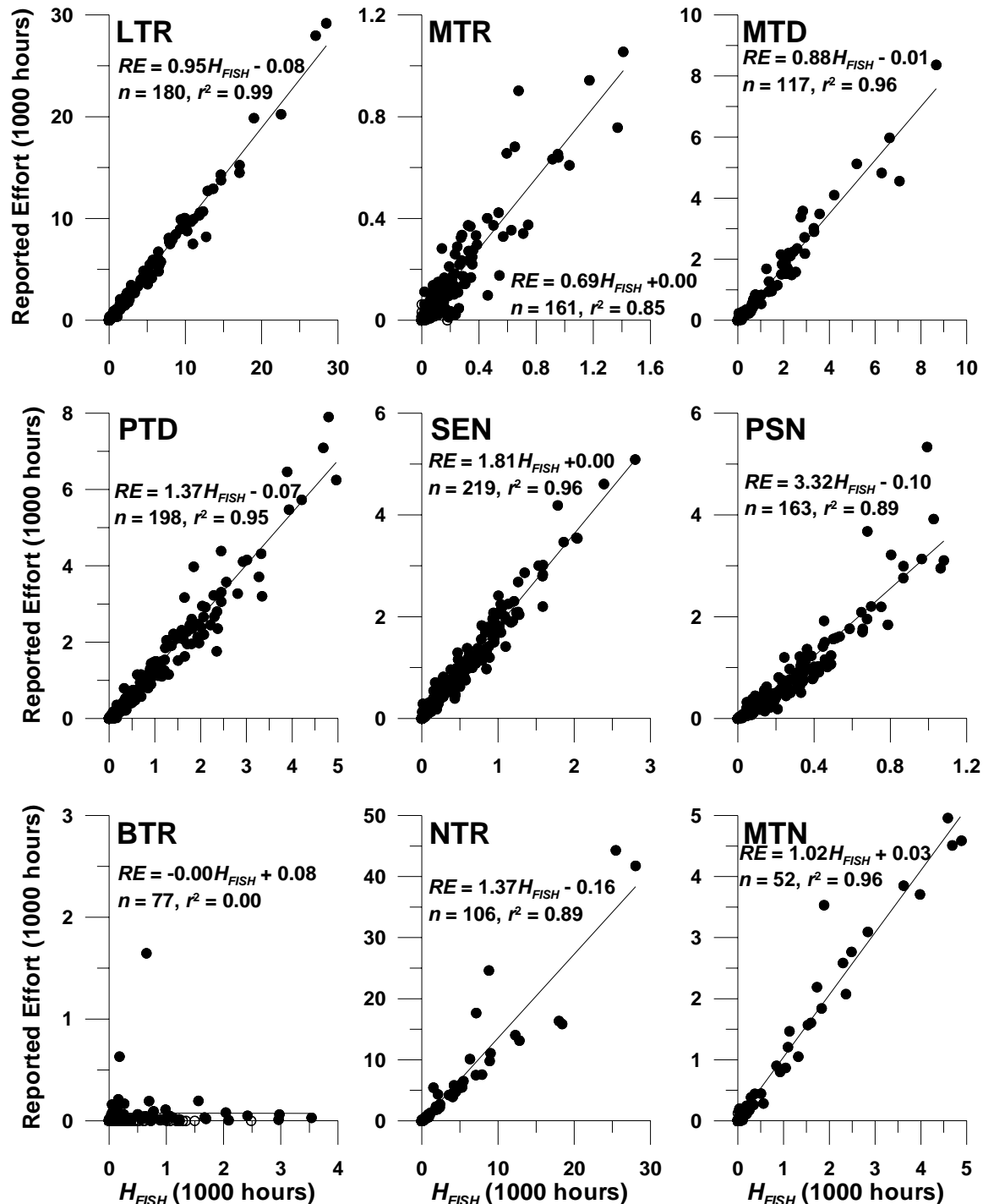


Figure 8.4.1.5. Validation of the fishing effort model for nine fishing gears used by Scottish fishermen in the North Sea. Plots show modelled hours-fishing against reported hours-fishing for each ICES rectangle in the two years 1997 and 1998 for each gear. Gear codes are explained in Table 8.4.1.2.

The largest deviations between the observed regression coefficient values and the expected value of one occurred for seine gear (SEN) and pair-Seine gear (PSN) (regression coefficients of 1.81 and 3.32 respectively). This reflected systematic issues associated with these two gears. Hours-fishing recorded by the observers for both seine gears included only the time when the gear was actually fishing, ie when it was being recovered. However, skippers reporting effort tended also to include the time when the gear was being set. Average haul duration in the observer data set was 1.6h, consistent with a net set of 13 coils, each 220m in length, on each side (Galbraith & Kynoch 1990). Adding the width of the net and including the bridles, an average tow of 1.6h would involve steaming a distance of 6km to lay the gear. Assuming a speed of 9.25 km h^{-1} (5 knots), setting the gear would take 0.65h on average. Thus typically, skippers would report 2.25h of effort for each 1.6h tow noted by the observers, a factor difference of 1.4. Taking this systematic difference into account would have produced a regression coefficient of around 1.29 for seine gear (SEN), much closer to a value of one. The same issue affects the pair-Seine data. However, the main reason why the pair-seine (PSN) regression coefficient is almost a factor of two higher than the single vessel seine gear coefficient resides in the fact that both vessels involved in the operation report their effort, whilst in our model we have split the effort between the two vessels (it is a single fishing operation taking a finite amount of time split between the two vessels). It is also likely that some pair-Seine activity was erroneously recorded as single vessel seine fishing in the FIN database. Since this would have involved two vessels, this again would have tended to inflate the SEN regression coefficient.

For the remaining gears, the deviations between the observed and expected regression coefficients were relatively minor. The double reporting of activity by both vessels in a paired-vessel activity almost certainly contributed to the slightly higher than anticipated regression coefficient for paired demersal trawling activity (PTD). The higher than expected *Nephrops* otter trawl (NTR) was mainly driven by two high effort values exerting considerable leverage. Generally the NTR data tended to fall close to a line with slope of one. For heavy otter trawl the coefficient was relatively low (0.69), suggesting that the model tended to over-estimate reported effort. The P_{FISH} parameter value for MTR, at 0.88, was the highest of any of the gears modelled. However, observer sample size for this gear, and hence our confidence in the parameter value, was least ($n=2$). A lower MTR value, closer to 0.65, that of LTR for example, would have resulted in a regression coefficient very close to one. By the late 1990s, MTR represented less than 3% of total otter trawling activity by Scottish fishing vessels. Thus the consequences for any assessment of the impact of Scottish otter trawling activity on the marine ecosystem and environment of an error in the MTR P_{FISH} parameter value of this sort of scale would have been negligible. For the time being therefore, we continued to use the value of 0.88 until additional observer data become available for analysis.

8.4.2. Modelling Dutch hours-fishing

The Dutch VIRIS database contains information on fishing activities of the entire Dutch fleet, which consists primarily of beam trawlers and otter trawlers. The database distinguishes different segments of each component of the fleet based on their engine-power. The Dutch beam trawl fleet primarily operates in the south-eastern North Sea targeting plaice (*Pleuronectes platessa*) and sole (*Solea vulgaris*). Two principal fishing units are distinguished within the Dutch beam trawl fleet; “Large vessels” with engine-power of ≥ 221 kW and “Euro cutters”, with engine power < 221 kW. These differ in their fishing practice and gear characteristics. Typically the large vessels deploy two 12 m beam trawls and are prohibited from fishing in the 12 nm coastal zone, or the “Plaice box”, whereas

the eurocutters deploy two 4m beam trawls but are allowed to fish inside this zone. The Dutch otter trawl fleet is considerably smaller than the beam trawl fleet. It principally targets cod, tending to operate in the south eastern and central North Sea. The VIRIS database stores the information recorded in individual fishing skipper's logbooks. This includes the date/time of the start and end of each fishing trip, thus enabling trip duration to be determined. The ICES rectangles fished, the type of gear used, and the landed catch of each fish species taken from each rectangle are also noted. The "days absent from port" (ie trip duration) for each fishing voyage was assigned to the ICES rectangles visited *pro-rata* with the landed catch from each rectangle.

A second, high resolution dataset, the APR/VMS database, consists of "Automated Position Registration" (APR) and "Vessel Monitoring through Satellite" (VMS) data. These are geographically referenced to a resolution of 1 minute latitude x 2 minute longitude (approximately 1x1 Nm). APR data were derived from a sample of about 10% of the Dutch beam trawl fleet that was equipped with APR equipment for the period 1993-2000 during which the position of the vessels was recorded every 6 minutes (Rijnsdorp 1998). The VMS data became available from 2000 onwards when positions of all EU vessels >24 m were recorded for enforcement purposes. From September 2003 onwards this was extended to vessels >18 m and subsequently from the 1st of January 2005 to vessels >15 m. Positions are recorded approximately every 2 hours. Although these data are collected by all EC countries for enforcement purposes, not all countries have access to VMS data for research purposes. For the Dutch beam trawl fleet VMS data from only a subset of the vessels (approximately 40%) are available for research purposes (Piet et al 2000). In addition to detailed data on track positions, some of the vessels provided data on a haul-by-haul (HBH) basis of the catch of the target species, the trawling speed and the times of shooting and hauling of the gear. These data allowed time actually fishing in each day a vessel was away from port to be determined. The larger beam trawler fished for 16.6h of each day, while the smaller "Euro cutters" fished for 17h in each day. This information allowed days-fishing per ICES rectangle to be converted to hours-fishing. A value of 15h per day was assumed for otter trawlers, similar to the P_{FISH} value for Scottish otter trawlers (see section 8.4.1).

8.4.3. Temporal variation in North Sea international fishing effort (MAFCONS participants)

Trends in annual fishing effort are shown disaggregated by main gear category for two time periods, 1990 to 1995 (Jennings et al 1999) and 1997 to 2004 determined from the MAFCONS database, and by country for the latter time period only (Figure 8.4.3.1). These data suggest that fishing activity in the North Sea peaked around 1994, and since then effort (in terms of hours fishing) has declined. This is particularly apparent for beam trawl, otter trawl directed at fish for human consumption, and seine gear. If anything, otter trawling for Nephrops has increased between 1997 and 2004. Data for the earlier time period could not be disaggregated to country, but this was possible for the latter period. Between 1997 and 2004, fishing effort by Norwegian registered vessels actually increased by approximately 19%, while effort by vessels registered to the four other countries all decreased over the same period. The most marked declines were apparent for UK registered fishing vessels (49% England, Wales and Northern Ireland and 40% Scotland). Effort by Dutch and German fishing vessels declined by 19% and 13% respectively.

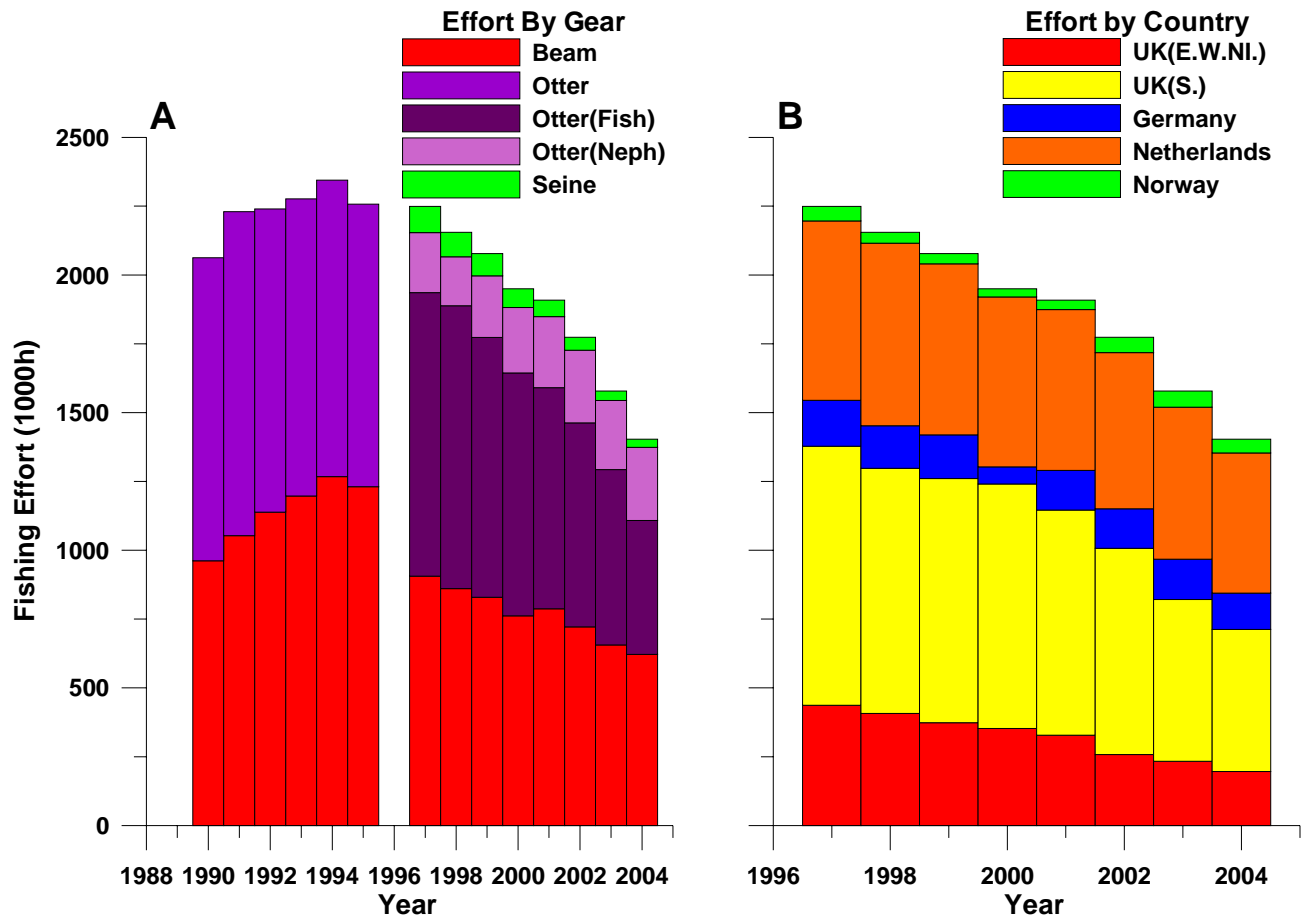


Figure 8.4.3.1. Trends in total annual fishing effort: (A) over the period 1990 to 2007 disaggregated by main gear category and including the data compiled by the earlier “Biodiversity” project (Jennings et al 1999), and (B) over the period 1997 to 2004 disaggregated by each of the countries contributing to the MAFCONS database. Data shown only for the countries that contributed to the respective databases, eg UK (England Wales and Northern Ireland), UK (Scotland), Denmark, Norway, Germany, and the Netherlands (1990 to 1995, Jennings et al 1999) and UK (England Wales and Northern Ireland), UK (Scotland), Norway, Germany, and the Netherlands (1997 to 2004, MAFCONS database).

8.4.4. Spatial variation in North Sea international fishing effort (MAFCONS participants)

Maps of the spatial distributions of average annual fishing effort were determined for four main fishing gear categories, beam trawl, otter trawl directed at fish for human consumption, otter trawl directed at Nephrops, and seine gear, covering two four year periods, 1997 to 2000 and 2001 to 2004 (Figure 8.4.4.1.1). Maps of average annual fishing effort for beam trawl and all otter trawl covering the period 1990 to 1995 (Jennings et al. 1999) are also included for comparison (Figure 8.4.4.1.1). For each of the four gears, the spatial distributions of effort were almost identical in each of the three time periods; any variation simply reflecting the reduction in effort overall over the time period (section 8.4.3). Beam trawling was primarily a southeastern North Sea activity, while otter trawling was principally a western North Sea activity, with two main patches of activity, one in the northwest and the second in the western central North Sea. Seine fishing was widespread across the

central and northern North Sea with patches of heaviest activity occurring in the northwest and northeast.

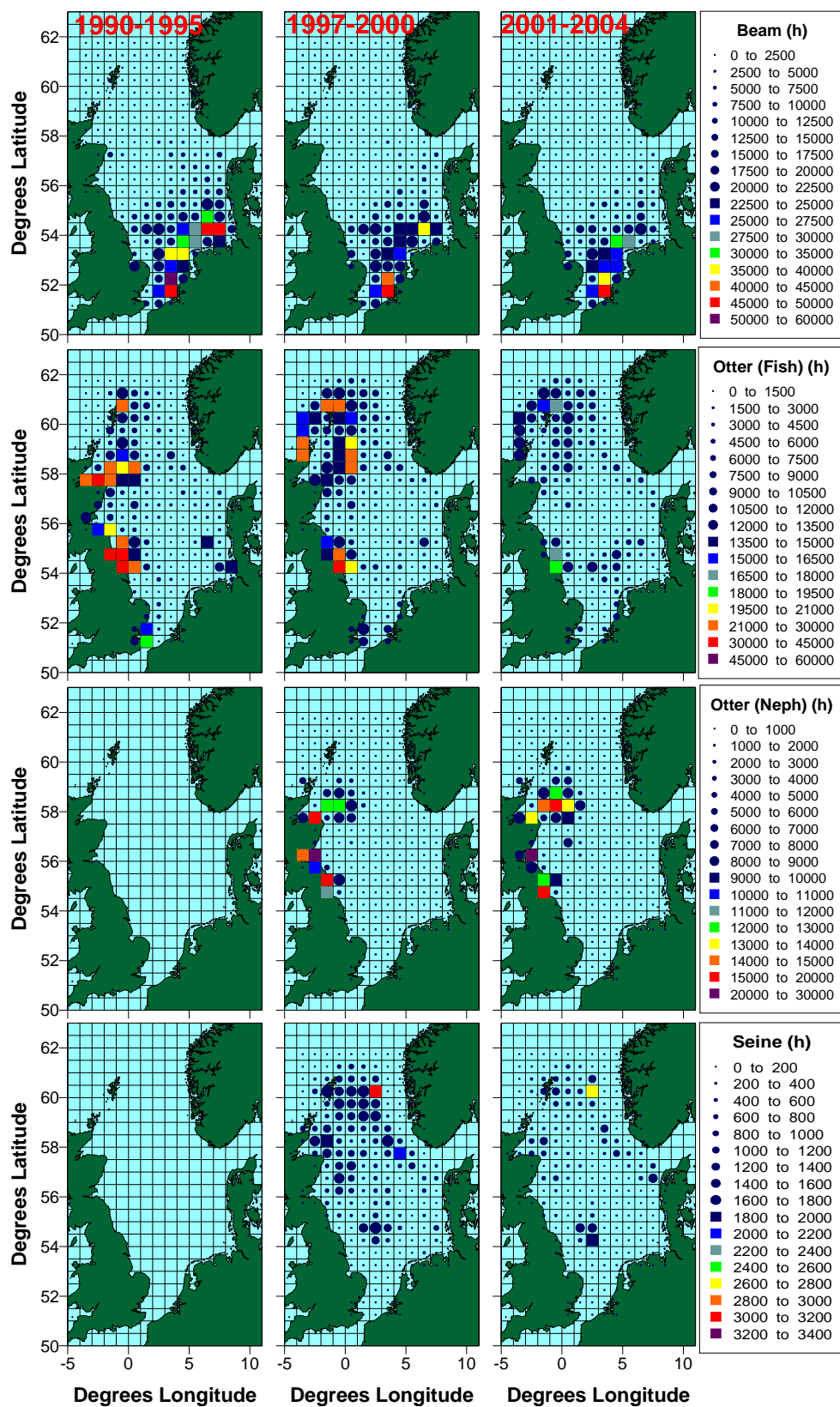


Figure 8.4.4.1.1. Distributions of average annual fishing effort by beam trawl, otter trawl directed at fish for human consumption {Otter (Fish)}, otter trawl directed at *Nephrops* {Otter (Neph)}, and seine gear covering the periods 1990 to 1995 (data from Jennings et al., 1999), 1997 to 2000, and 2001 to 2004.

Maps of the spatial distributions of average annual fishing effort were determined for the five countries contributing to the MAFCONS database, UK (England, Wales and Northern Ireland), UK (Scotland), Germany, the Netherlands, and Norway, covering two four year periods, 1997 to 2000 and 2001 to 2004 (Figure 8.4.4.1.2). Although vessels from each of the five MAFCONS countries ranged widely across most of the North Sea, it was clear that fishing vessels from each country concentrated their activities in waters close to their national coastline.

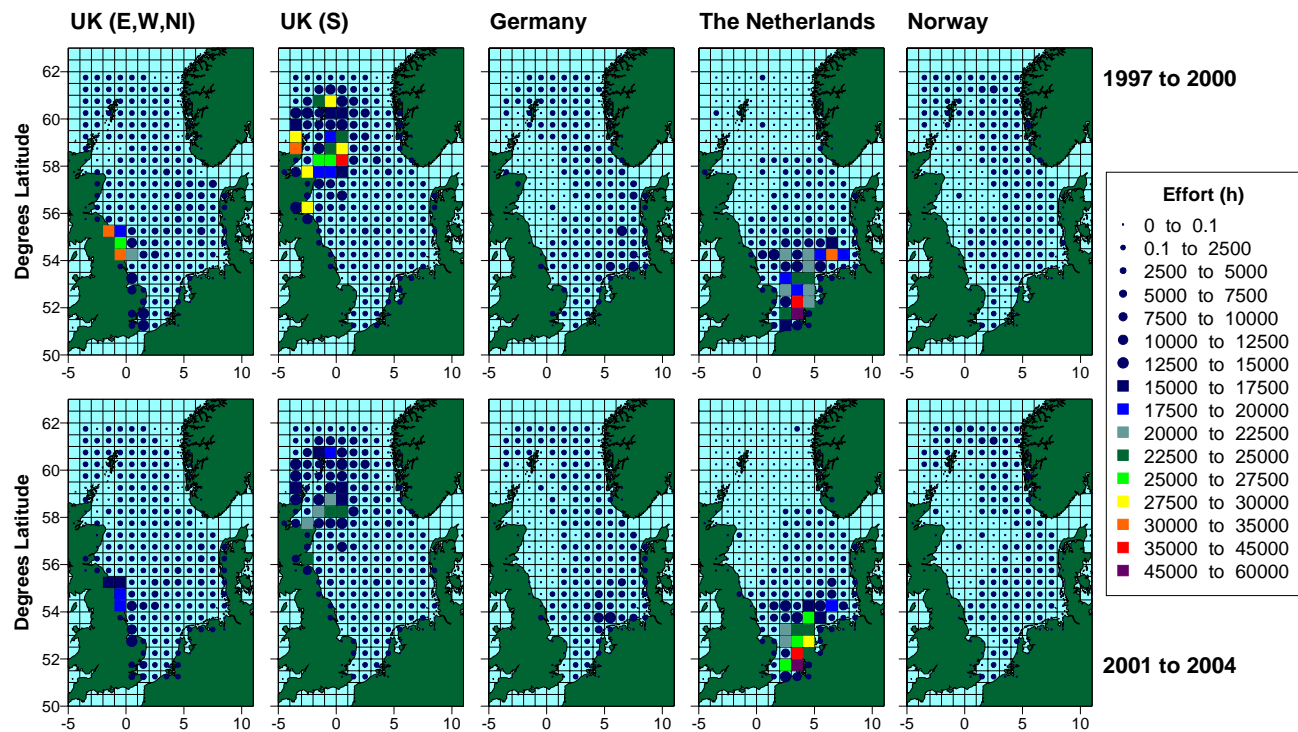


Figure 8.4.4.1.2. Distributions of average annual fishing effort by vessels registered to the UK (England, Wales and Northern Ireland), the UK (Scotland), Germany, the Netherlands, and Norway, in two periods, 1997 to 2000, and 2001 to 2004.

8.5. Modelling landings and effort by non-MAFCONS countries

8.5.1. Fleet composition of the non-MAFCONS countries

Temporal trends in the annual landings of each species by each of the countries operating significant fisheries in the North Sea were presented in section 8.3.1 (Figure 8.3.1.2). These data are summarised in Table 8.5.1.1, which gives the mean percentage of total annual North Sea landings of each species landed by each country over the period 1997 to 2004. Of the countries not participating in the MAFCONS project, this table suggests that, at a North Sea scale, Belgium landed significant quantities of sole, plaice and cod, Denmark landed significant quantities of cod, plaice and *Nephrops*, and France landed significant quantities of saithe and whiting. In terms of total North Sea landings, Swedish catches were relatively minor, but with saithe, haddock and whiting being the most important. Table 8.5.1.2 considers the landings from the perspective of each of the individual countries, giving the mean percentage contributed by each of the species to each country's total annual landings over the period 1997 to 2004. This table confirms that in tonnage terms: plaice, cod

and sole were the most important species for Belgian fishermen; plaice, cod and saithe were the main targets of Danish fishermen; saithe and whiting were the principal targets of French fishermen; and saithe, haddock and cod were of greatest importance to Swedish vessels.

Species	UKE	UKS	GER	NLD	NOR	BEL	DNK	FRA	SWD
Cod	10.59	31.83	5.57	10.02	10.36	5.48	21.39	2.58	0.96
Haddock	4.71	78.20	1.85	0.43	4.85	0.92	5.11	1.35	1.05
Whiting	8.92	61.33	1.77	10.40	0.26	1.82	0.48	15.31	0.03
Saithe	1.74	5.69	10.05	0.01	51.82	0.13	4.96	18.42	1.79
Plaice	13.69	10.58	5.14	41.90	2.02	7.41	17.40	0.58	0.00
Sole	2.83	1.73	4.69	69.28	0.17	8.71	3.90	1.62	0.00
<i>Nephrops</i>	14.32	67.19	0.00	3.68	0.71	2.38	11.71	0.00	0.00

Table 8.5.1.1. Mean percentage (calculated over 8 years, 1997 to 2004) of total annual North Sea landings of cod, haddock, whiting, saithe, plaice, sole, and *Nephrops* landed by each of the nine countries with significant North Sea fisheries; UK (England, Wales, and Northern Ireland) (UKE), UK (Scotland) (UKS), Germany (GER), the Netherlands (NLD), Norway (NOR), Belgium (BEL), Denmark (DNK), France (FRA), and Sweden (SWD).

Species	UKE	UKS	GER	NLD	NOR	BEL	DNK	FRA	SWD
Cod	24.38	18.41	16.09	11.60	8.72	26.31	34.51	7.35	17.93
Haddock	10.02	43.77	5.10	0.44	4.26	4.24	7.58	2.86	18.90
Whiting	6.50	12.13	1.53	3.56	0.08	2.95	0.23	11.65	0.17
Saithe	6.72	6.10	52.42	0.02	84.34	1.00	14.44	75.01	62.88
Plaice	40.76	8.19	19.94	58.05	2.37	47.77	35.95	1.76	0.11
Sole	2.30	0.36	4.92	25.23	0.05	14.67	2.11	1.36	0.01
<i>Nephrops</i>	9.31	11.04	0.00	1.09	0.18	3.06	5.17	0.00	0.00

Table 8.5.1.2. Mean percentage (calculated over 8 years, 1997 to 2004) of fish landed annually by each of the nine countries with significant North Sea fisheries, UK (England, Wales, and Northern Ireland) (UKE), UK (Scotland) (UKS), Germany (GER), the Netherlands (NLD), Norway (NOR), Belgium (BEL), Denmark (DNK), France (FRA), and Sweden (SWD), that consisted of cod, haddock, whiting, saithe, plaice, sole, and *Nephrops*.

Table 8.5.1.3 gives the number of vessels in the Belgian, Danish, French and Swedish fleets registered as using particular types of demersal fishing gear (Europa 2006). Beam trawlers made up 93% of the Belgian fleet. Figure 8.3.3.2.1 indicates that significant quantities of cod were caught by beam trawlers operating in the southeastern North Sea. This was also confirmed by the analysis of catch-effort relationships for the Dutch beam trawl fleet in Chapter 12. Given that 89% of Belgian landings consisted of plaice, sole and cod, and that appreciable quantities of whiting, and haddock would also be caught as a bycatch in beam trawls (Figure 8.3.3.2.1), it seems likely that almost all Belgian fishing activity in the North Sea consisted of beam trawling. This was indicated by the limited amount of Belgian fishing effort data available to the MAFCONS consortium derived from reports from vessels landing into English and German ports (9414 hours over the eight year period), which suggested that over 99.9% of Belgian fishing activity consisted of beam trawling. All activity by Swedish vessel in the North Sea would have consisted of otter trawling (Table 8.5.1.3), and again this is entirely consistent with their main target species saithe, haddock and cod, which are primarily caught in otter trawls (Figure 8.3.1.3, Chapter 12). Saithe and whiting were the main targets of the French fishing fleet in the North Sea. These species are primarily caught by otter trawl, and given

the relative numbers of otter trawlers and beam trawlers in the French fleet (Table 8.5.1.3), it is most likely that the vast majority of French fishing vessels operating in the North Sea were otter trawlers.

Fishing Gear	Belgium	Denmark	France	Sweden
Beam trawl	112	22	50	0
Otter trawl	8	612	1347	239
Twin otter trawl	0	0	6	0
Pair trawl	1	1	7	50
Danish seine	0	88	6	3
Scottish seine	0	0	0	0
Pair seine	0	0	1	0

Table 8.5.1.3. Numbers of Danish, Belgian, French and Swedish fishing vessels registered as using particular demersal fishing gear (Europa 2006).

Interpretation of the vessel composition data is less easy for the Danish fleet. Many of the 612 Danish otter trawlers would have been occupied in industrial fishing for sandeels and Norway pout, both of which involve otter trawling close to, if not in contact with, the seabed. This type of fishing activity has so far not been considered in detail in this study because Denmark is the principal country with quota for industrial fish (ICES 2005), and unfortunately no Danish institute was included in the MAFCONS consortium. Industrial fishing activity will be considered separately at the end of this section. For now we consider the amount, type, and distribution of Danish fishing activity for the main commercial fish for human consumption and *Nephrops*. The fact that plaice, cod and saithe are the primary target species of Danish fishing vessels suggests that their fleet operating in the North Sea consists of both otter and beam trawlers. However, the Danish fleet also includes a substantial number of seiners (Table 8.5.1.3). Again a limited amount of Danish effort data was available from vessels landing in English and German ports. Of the 5214 hours reported over the eight year period, 32% consisted of otter trawl effort directed at fish for human consumption, 4% of otter trawl directed at *Nephrops*, and the remainder, 64%, of seine fishing (Table 8.5.1.4). This confirmed the importance of Danish seine fishing activity, and perhaps suggests that the amount of Danish beam trawl activity taking place in the North Sea was relatively small. Alternatively, Danish beam trawlers may simply not have landed into English or German ports. These records, albeit limited in number, suggest that Danish seiners may have been responsible for much of the Danish plaice landings, again suggesting that Danish beam trawlers operating in the North Sea may be limited in number (Table 8.5.1.4). It must be stressed that the data in Table 8.5.1.4 are only “indicative”, and not representative of the Danish fishing fleet, which presumably lands primarily into Danish ports. In conclusion it seems likely that the Danish fleet operating in the North Sea must use all four of our main gear categories. However, given that the fleet is dominated by otter trawlers (Table 8.5.1.3), it seems most likely that otter trawlers, principally but not exclusively targeting fish for human consumption (Table 8.5.1.4), constitute the largest fraction of Danish fishing activity

Species	Otter(Fish)	Otter(Neph)	Seine
Cod	2.842	1.506	395.952
Haddock	27.732	12.252	4.870
Whiting	0.536	2.960	0.000
Saithe	0.078	0.000	3.428

Plaice	93.080	11.862	313.246
Sole	0.004	0.044	0.086
<i>Nephrops</i>	0.018	0.552	0.000
Effort (h)	1692	204	3318

Table 8.5.1.4. Landings into English and German ports of cod, haddock, whiting, saithe, plaice, sole and *Nephrops* by Danish fishing vessels using otter trawl directed at fish for human consumption (Otter(Fish)), otter trawl directed at *Nephrops* (Otter(Neph)), and seine gear. The hours of fishing effort reported for each gear are also indicated.

8.5.2. Modelling landings and effort by Belgian, Swedish, French and Danish fleets

For each of the modelled fleets, or sub-fleet components, total reported landings (or a fraction of total reported landings if dealing with sub-fleet components) of the targeted species (ICES 2005) were distributed *pro rata* with landings of the same species by the same gear over the area of operation by all five MAFCONS countries combined. *Cpue* of the species concerned in each rectangle by the MAFCONS countries was then used to estimate the fishing effort required in each rectangle by the modelled fleet (or sub-fleet component) to take these landings. Given these estimates of fishing effort by a particular gear in each rectangle by the modelled fleet (or sub-fleet component), *cpue* for each of the remaining six species in each rectangle by the combined MAFCONS countries was used to estimate the bycatches of these species associated with the modelled pattern of fishing effort. For each of the four countries' modelled fishing fleets, this process was repeated in an iterative fashion, manipulating the area of operation by each fleet (or sub-fleet component) and altering the proportion of total reported landings of the targeted species assigned to the fleet (or sub fleet component) until the best fit to the total reported landings of each species by each country (ICES 2005) could be achieved.

8.5.2.1. *Belgian fishing fleet strategy*

In section 8.5.1 we conclude that the Belgian fleet consists almost entirely of beam trawlers targeting the two flat fish species and catching their quotas of cod, and other species, as a bycatch. Furthermore, like the Dutch beam trawl fleet (see Chapter 12), we assume that Belgian beam trawl activity is primarily influenced by their pursuit of sole, the more valuable of the two flatfish. We also assume that, as indicated by the effort patterns by country illustrated in Figure 8.4.4.1.2, the Belgian beam trawl fleet would preferentially have operated reasonably close to Belgian ports. Consequently we limit their activity to ICES rectangles south of latitude 55°N. Belgian beam trawling alone was insufficient to account for the Belgian landings of the other six species. Thus, either additional beam trawling directed at plaice was required, or some Belgian otter trawling also occurred. This latter possibility was examined first.

We again assumed that Belgian otter trawlers would tend to operate as close to the Belgian coast as possible, but in this case we assumed that they would have to travel further north in order to reach suitable otter trawling grounds. Thus, Belgian otter trawling was limited to the area south of latitude 58°N. We also assumed that the otter trawlers would be primarily targeting cod. Initially we estimated the amount of otter trawl effort required to take the cod landings remaining after accounting for the cod bycatch in the beam trawl fishery. However, it quickly became apparent that the amount of effort required, between 59,000h.y⁻¹ and 104,000h.y⁻¹, was not possible given the size of the Belgian otter trawl fleet. A fleet of nine vessels could manage a total of approximately

30,300h.y⁻¹ (each vessel fishing 0.65 of the working day (see Table 8.4.1.7), fishing for 18 days in each month). We also assumed that the Belgian otter trawl fleet reduced steadily in size by 33% over the period 1997 to 2004, in line with decommissioning and the reduction in cod quotas. Consequently total annual Belgian otter trawl effort at the start of the period was assumed to be 40,450h.y⁻¹. These estimates of annual Belgian otter trawl effort were then distributed across the Belgian region pro rata with otter trawl effort for the five MAFCONS countries. Assuming the same otter trawl *cpue* for each species in each rectangle in each year as for the MAFCONS countries, landings of each species by year and rectangle by Belgian otter trawlers could be estimated. Belgian landings of all species estimated in this way satisfied the Belgian annual landings reported by ACFM (ICES 2005), although modelled cod landings were slightly lower than total reported Belgian landings in each year. Landings of plaice by otter trawl were sufficient to make up the shortfall in this species after estimating Belgian beam trawl effort. Any discrepancies between estimated Belgian landings and landings reported by ACFM (ICES 2005) were relatively small and easily coped with by assuming quota swapping between countries and discarding by Belgian fishers.

8.5.2.2. *Swedish fishing fleet strategy*

In section 8.5.1, we conclude that the Swedish demersal fleet consists entirely of otter trawlers principally targeting saithe. We assume that these vessels would have to travel long distances to reach the main saithe grounds along the shelf-break at the northern boundary of the North Sea and along the Norwegian Deep, so no spatial restrictions were placed on their fishing activity. Swedish otter trawling directed at saithe in this way also accounted for all the Swedish landings of the remaining six species; slightly more in some years and less in others. Whiting landings were considerably in excess of the Swedish quota in ratio terms, because their quota was so small. But in terms of absolute biomass, the excess, at between 380t and 950t, would probably have been discarded, landed in excess of quota, or quota swapped with another country. Swedish landings of plaice were also orders of magnitude greater than their quota, but as for whiting this was primarily because the quota was so small. Again in terms of actual biomass, the over-catch of between 100t and 220t would either have been discarded, landed in excess of quota, or quota swapped with another country.

8.5.2.3. *French fishing fleet strategy*

If, like the Swedish otter trawlers targeting saithe, the French otter trawl fleet was allowed to roam freely so that French otter trawl effort was widespread and sufficient to catch reported total French saithe landings (ICES 2005) from all possible North Sea rectangles, then the modelled bycatch of several other species, but particularly of haddock, far exceeded reported French landings. Consequently a different approach was adopted for the French otter trawl fleet. Saithe *cpue* for the five MAFCONS countries otter trawl fleets was consistently high (>0.1t.hr⁻¹ in five or more years) in 37 ICES rectangles situated in the far north and northeastern North Sea. These 37 rectangles accounted for between 90% and 97% of total saithe landings by the five MAFCONS countries. We assume therefore that the French otter trawl fleet, primarily targeting saithe, restricted their activities to these 37 rectangles. Total bycatch of other species from these 37 rectangles was now considerably lower, but still sufficient to result in an over-catch of haddock of between 1100t and 2900t each year. Even restricted to the 37 most productive saithe rectangles, French otter trawl haddock bycatches were between two and six times total French reported landings for this species each year. Because haddock *cpue* was also high in almost all of the North Sea where high saithe *cpue* occurred,

little could be done to reduce the size of this haddock bycatch unless French otter trawling for saithe was restricted to an unrealistically small number of ICES statistical rectangles. We assume that this large bycatch is therefore “real”, and that these fish were either discarded, landed over quota, or dealt with through quota swaps with other countries. Although not so extreme a similar situation was apparent for cod. Every year from 1999 onwards the cod bycatch in this targeted saithe fishery exceeded total reported French cod landings by factors of between 1.2 and 3.6 times, amounting to over-catches of between 500 and 2500t. Again this was considered to be the “real” situation. In 1997 and 1998, French saithe landings were considerably below their quota (Chapter 12), but in 1999 this situation changed and French saithe landings increased markedly, remaining relatively high thereafter. At this time cod quotas fell markedly as the cod stock recovery plan took effect. However, despite reduced cod stocks and an associated reduction in cod *cpue*, increased French effort in their otter trawl fishery for saithe quickly resulted in the cod bycatch exceeding the reduced quota. We assume these fish were either discarded, landed over quota, or landed through quota swapping. Whiting bycatch in the saithe otter trawl fishery represented between 12% and 41% of the French quota, with shortfalls of between 1100t and 3700t each year. Either some small amount of French otter trawling directed specifically at whiting took place somewhere in the southern North Sea where whiting *cpue* was particularly high, or this unfished quota was swapped to allow some of the excess haddock and cod to be landed. It was difficult to model a directed whiting fishery without seriously increasing the size of the cod bycatch excess. Doing so would have implied even higher levels of French cod discarding, or over-quota landing, and we therefore assumed the latter situation. The French quota for *Nephrops* is around 25t.yr⁻¹ and modelled landings in the saithe otter trawl fishery bycatch varied between 4t and 41t each year.

Landings of plaice from the saithe otter trawl fishery, at between 20t and 42t each year, represented between 4% and 10% of total reported French landings. Almost no sole were landed in this fishery. We assume that of these two flatfish species, sole were primarily targeted by the French beam trawl fleet. If the French beam trawl fleet was allowed to operate with unlimited access to all the North Sea, then the beam trawl effort required to take French sole landings would have resulted in a massive over-catch of plaice. So again French beam trawlers had to be restricted to ICES rectangles where sole *cpue* was relatively high and plaice *cpue* relatively low. This was achieved by restricting the French beam trawl fleet to ICES rectangles south of latitude 55°30'N. Within this region 15 ICES rectangles had consistently high sole *cpue* (>0.017t.h⁻¹ for five or more years), while only one rectangle had consistently high plaice *cpue* (>0.08t.h⁻¹ for five or more years). In five of the eight years, plaice bycatches in the sole beam trawl fishery and saithe otter trawl fisheries combined were between 0.93 and 1.20 times total reported French plaice landings (ICES 2005). In the three years, 1998, 2001 and 2004, modelled plaice over-catches of 391t, 291t, and 188t respectively represented 1.8 times total reported French plaice landings for these years. Modelled bycatches of cod in 1997 and 1998 of 51t and 283t respectively, even when combined with the bycatch from the saithe otter trawl fishery, were still not sufficient to exceed total reported French landings. In the six years when the modelled saithe otter trawl fishery cod bycatch exceed total reported French landings, a further 25t to 130t were expected to have been caught in the modelled beam trawl fishery; numbers that were too small to alter in any significant way our conclusion stated above. Between 16t and 90t of whiting were expected to have been taken each year in the sole beam trawl fishery. Total combined modelled whiting landings in both the saithe otter trawl and sole beam trawl fisheries remained well below total French reported whiting landings (ICES 2005), and the comment made above therefore still holds true. Modelled landings of saithe, haddock and *Nephrops* in the French beam trawl fishery were negligible (<1t) in all years.

8.5.2.4. *Danish fishing fleet strategy*

In section 8.5.1. we conclude that the Danish demersal fleet consists primarily of otter trawlers, but also contains significant numbers of beam trawlers and seiners. We assume that the otter trawlers would principally target saithe and cod, the beam trawlers plaice, and seiners cod. The Danish fleet was therefore more complicated to model. Danish trawlers reported relatively high landings of cod and saithe, yet reported landings of haddock and whiting were relatively low (ICES 2005). In a mixed fisheries situation it proved impossible to model the reported Danish landings of cod and saithe without large over-catches of haddock and whiting being taken. After many iterations the best fit was achieved assuming five components to the Danish fleet, each with restricted areas of operation. Three otter trawl components were necessary; one specifically targeting saithe in the same 37 ICES rectangles in the north and northeastern North Sea fished by the French otter trawl fleet, the second specifically targeting cod and operating within an area of the North Sea bounded by latitudes 56°N to the south, 59.5°N to the north and 1°E to the west and within this area avoiding the rectangles fished by trawlers targeting saithe, and the third component targeting *Nephrops* and restricting their activities solely to the *Nephrops* grounds in four ICES rectangles in the Botney Cut (*Nephrops* management area H, functional unit 5) and three ICES rectangles off the Horn Reef (*Nephrops* management area H, functional unit 33) regions of the North Sea. A small seine gear fleet was also assumed to operate within the same region as the otter trawl component targeting cod. This strategy provided the best compromise between achieving reported cod, saithe and *Nephrops* landings, while keeping haddock and whiting over-catches to levels compatible with discarding, quota swapping, and over-quota landings. Finally, the Danish fleet also included a small beam trawl component principally targeting plaice, which was required to operate north of latitude 54°N to avoid excessive bycatch of sole.

8.5.3. Combined landings and effort by the non-MAFCONS countries

8.5.3.1. *Landings*

Just as earlier we compared landings data submitted by each MAFCONS country to landings data reported by ICES for these same countries (ICES 2005), here we compare modelled landings for the four non-MAFCONS fleets, with landings reported by ICES (2005) for Belgium, Sweden, France and Denmark (Figure 8.5.3.1.1). It is immediately apparent that for some species, the fits are not so good (compare Figure 8.5.3.1.1 with Figure 8.3.1). However, modelled landings data for the four fleets also includes all the fish caught over-quota that would either have been landed (quota swapping, illegal landings) or discarded; fish that were excluded from the comparisons shown in Figure 8.3.1. Agreement between modelled and reported landings were close for the four species considered to be primarily targeted by the four non-MAFCONS fleets; cod, saithe, sole and *Nephrops*. Modelled landings of haddock were primarily driven by Danish and French fleets targeting relatively large saithe quotas. Landings of saithe increased over the 8 year period, yet reported landings of haddock supposedly declined (Figure 8.3.1.3). The model was unable to replicate this. In order to achieve reported saithe landings haddock catches also increased over the period, yet reported landings theoretically declined. The situation for whiting was similar. Again reported landings declined markedly over the period (Figure 8.3.1.3), yet modelled bycatches in the saithe and cod directed otter trawl fisheries failed to reflect this. Despite restricting the modelled beam trawl fleets to areas where sole cpue was highest, plaice bycatch in the modelled landings for

the non-MAFCONS fleets consistently exceeded reported landings, but by an amount that was consistent with discards:landings ratios (REFERENCE).

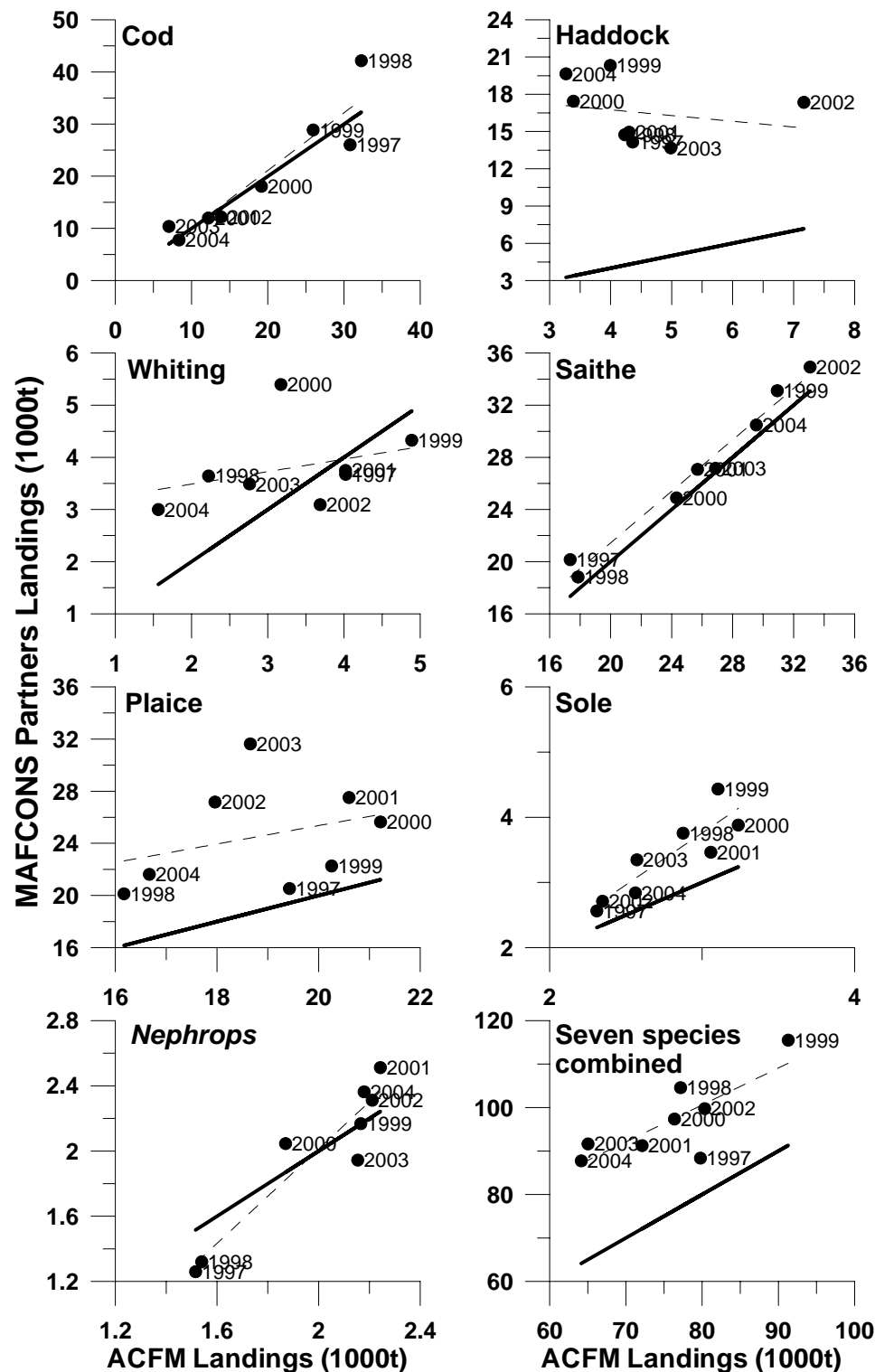


Figure 8.5.3.1.1. Correspondence between the landings data analysed to determine spatial patterns in landings as part of the MAFCONS project and the data reported to ICES for the stock assessments. Plots show total annual landings of each species, and all seven species combined, derived from the ACFM report (ICES 2005) summarising the ICES advice

following the 2005 stock assessment process and annual summed modelled landings data for the four countries not involved in the MAFCONS project. Solid lines indicate relationship expected for perfect agreement and dashed lines show linear fits to the data.

Modelled landings of each species by gear for the four non-MAFCONS fleets (Figure 8.5.3.1.2) showed similar patterns to those determined from data reported by the five MAFCON countries (Figure 8.3.1.3). The four gadoid species were primarily caught by otter trawl, although a slightly greater proportion of cod and whiting were taken in the modelled otter trawl targeting *Nephrops*. Sole and plaice were taken primarily by beam trawlers, although the proportion of plaice taken by modelled otter trawl was slightly higher. *Nephrops* was taken almost entirely by otter trawl.

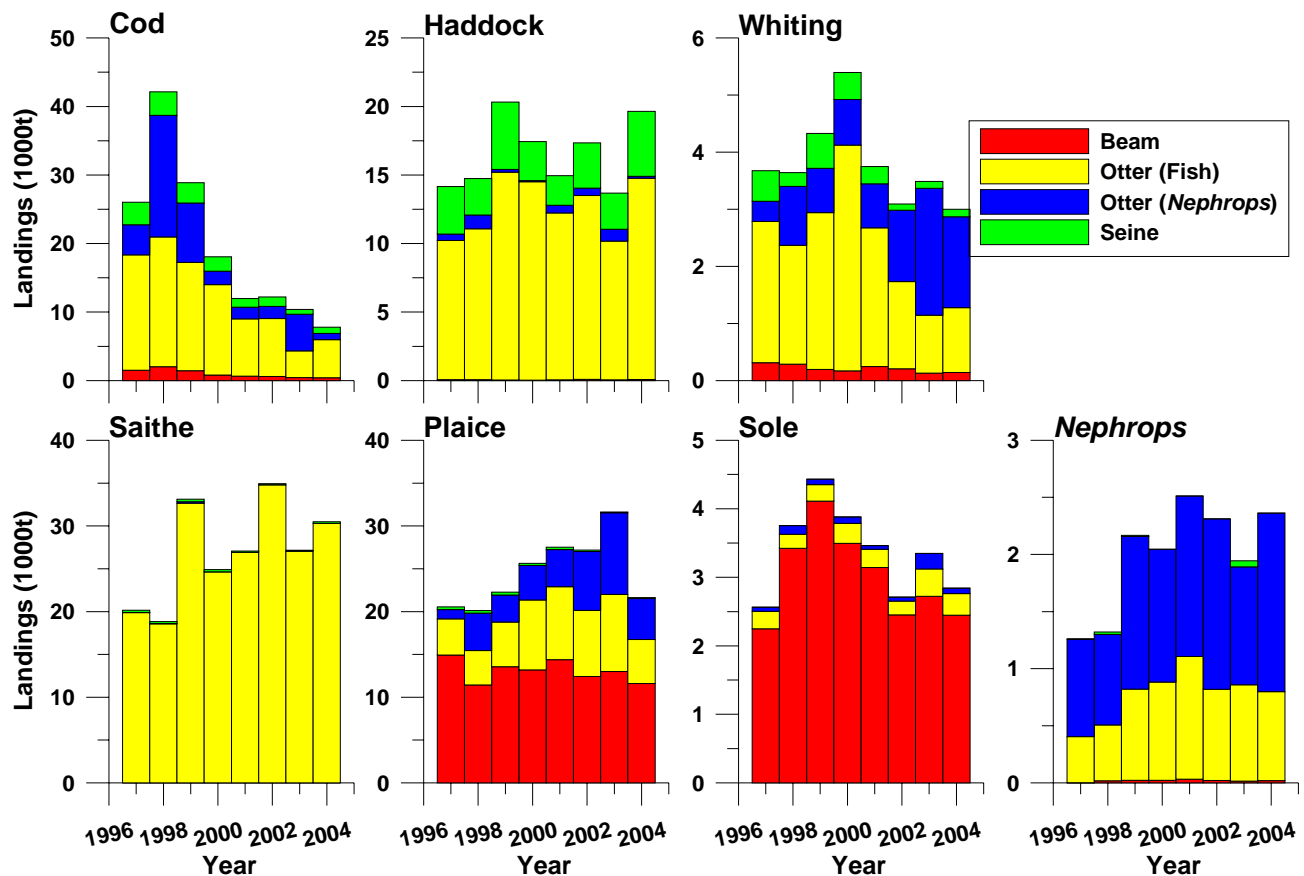


Figure 8.5.3.1.2. Trends in the landings of cod, haddock, whiting, saithe, plaice, sole, and *Nephrops* landed by the countries not participating in the MAFCONS project (MAFCONS landings database) indicating the amounts caught in four major gear categories; beam trawl, otter trawl directed at fish for human consumption, otter trawl directed at *Nephrops* and seine gear.

8.5.3.1.1. Spatial trends in landings

Spatial distributions of modelled landings of each species in all gears combined in each year by the four non-MAFCONS countries combined (Figures 8.5.3.1.1.1 to 8.5.3.1.1.7) were broadly similar to plotted for reported landings by the five MAFCONS countries (see Figures 8.3.2.1.1 to 8.3.2.1.7). The same changes in the spatial distributions of landings over time were in general also apparent. However, for the gadoid species in particular, the spatial restrictions imposed on the modelled fleets

were clearly apparent (Figures 8.5.3.1.1.1 to 8.5.3.1.1.7). However, since the modelled fleets all operated from countries to the south of the North Sea, or to the east, these spatial patterns appear entirely reasonable.

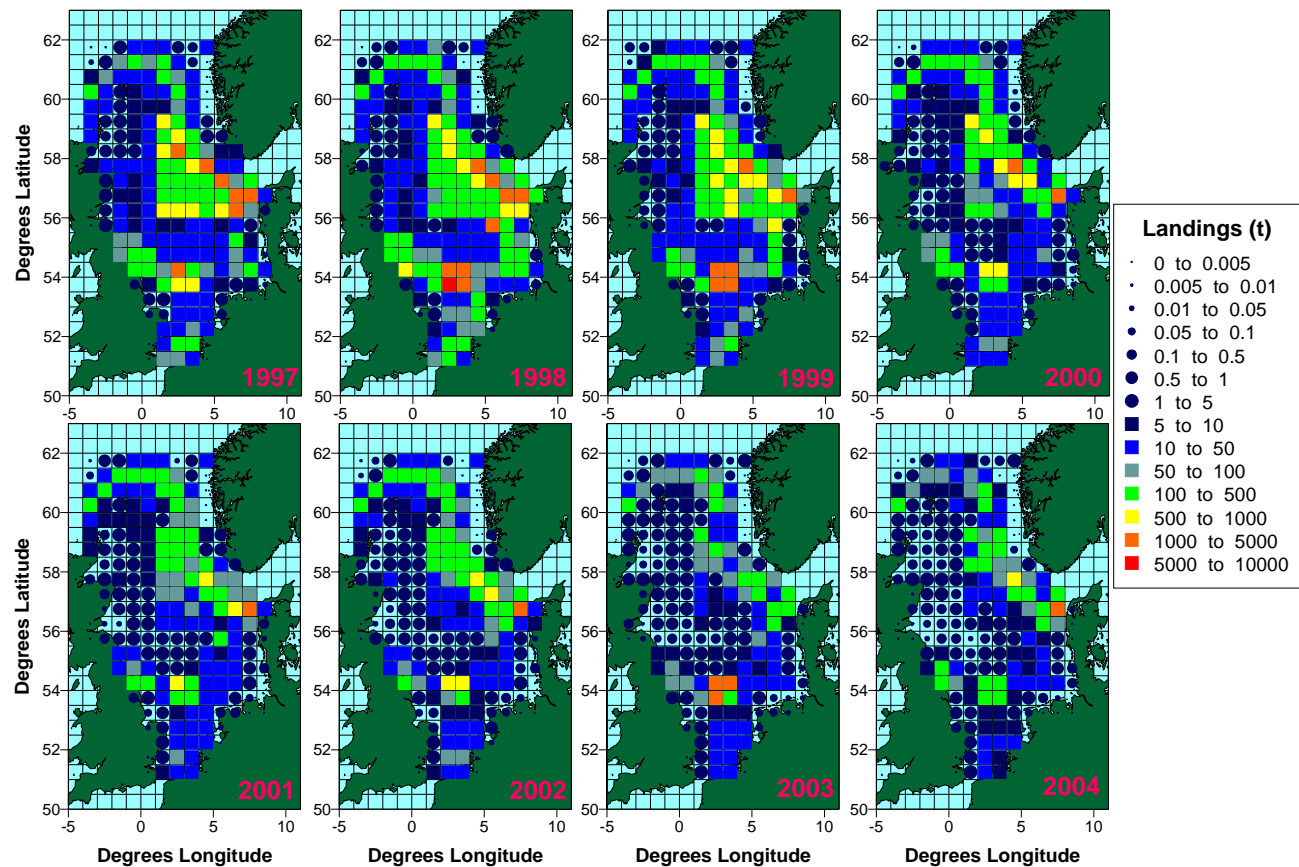


Figure 8.5.3.1.1.1. Spatial distributions of cod landings across all fishing gears by countries not participating in the MAFCONS project in each of the years between 1997 and 2004.

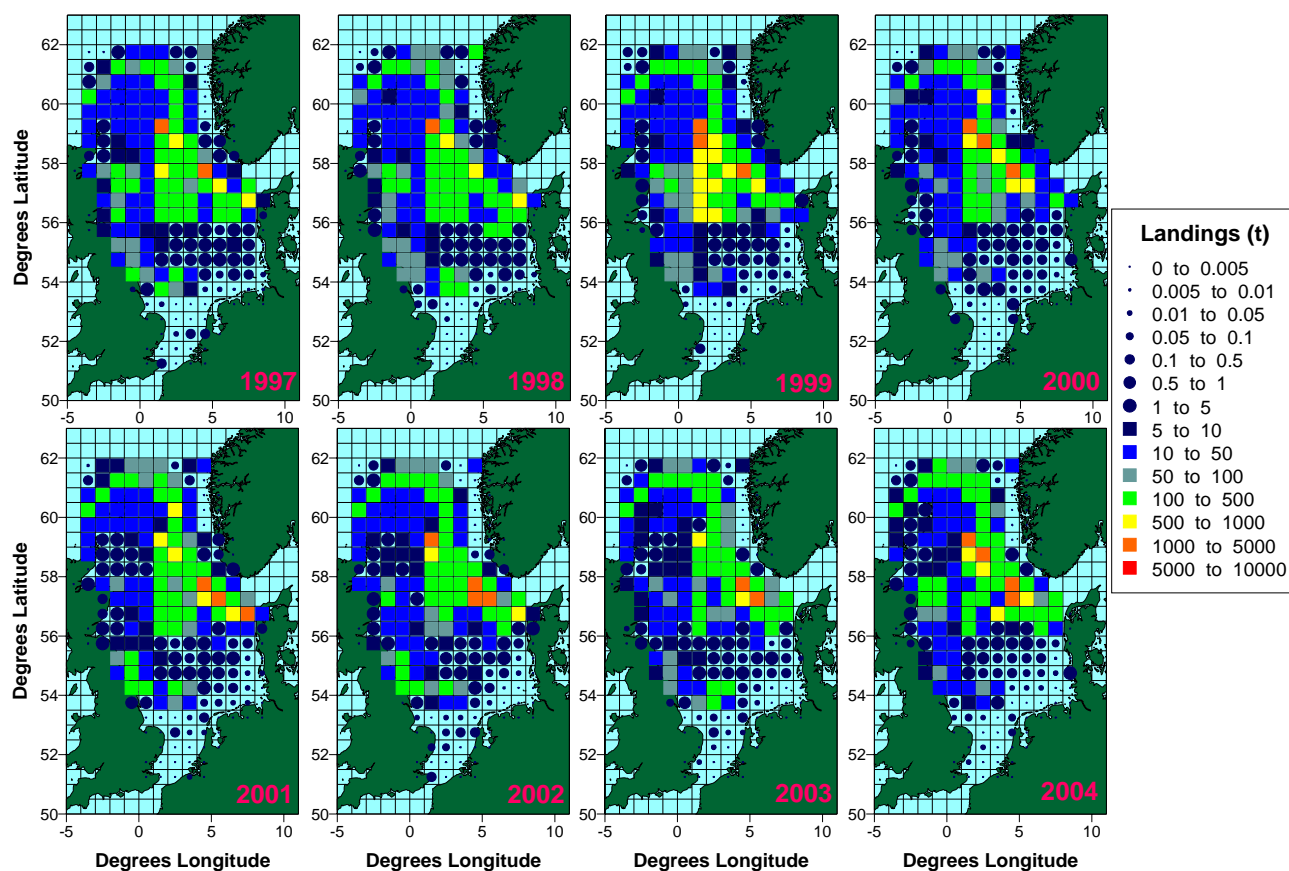


Figure 8.5.3.1.1.2. Spatial distributions of haddock landings across all fishing gears by countries not participating in the MAFCONS project in each of the years between 1997 and 2004.

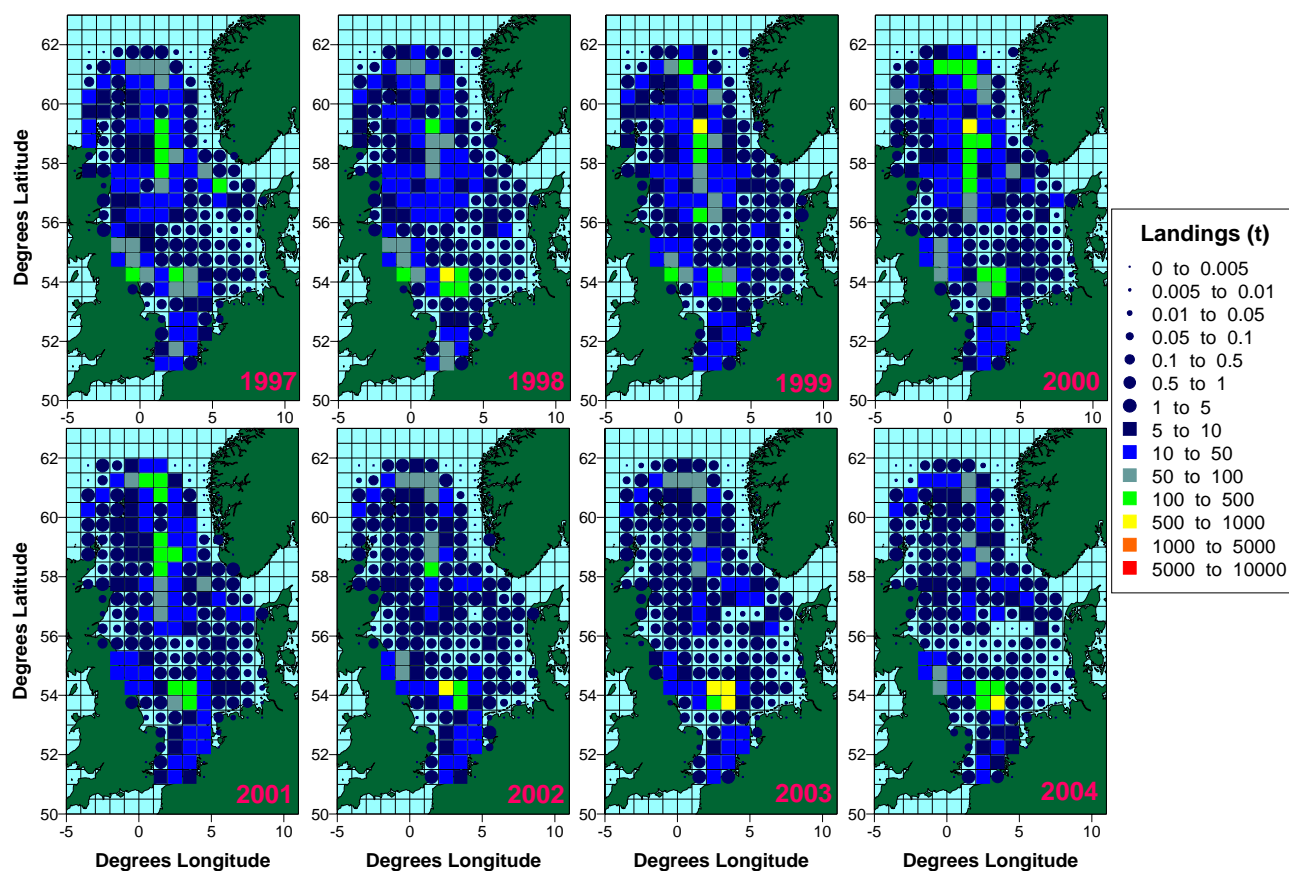


Figure 8.5.3.1.1.3. Spatial distributions of whiting landings across all fishing gears by countries not participating in the MAFCONS project in each of the years between 1997 and 2004.

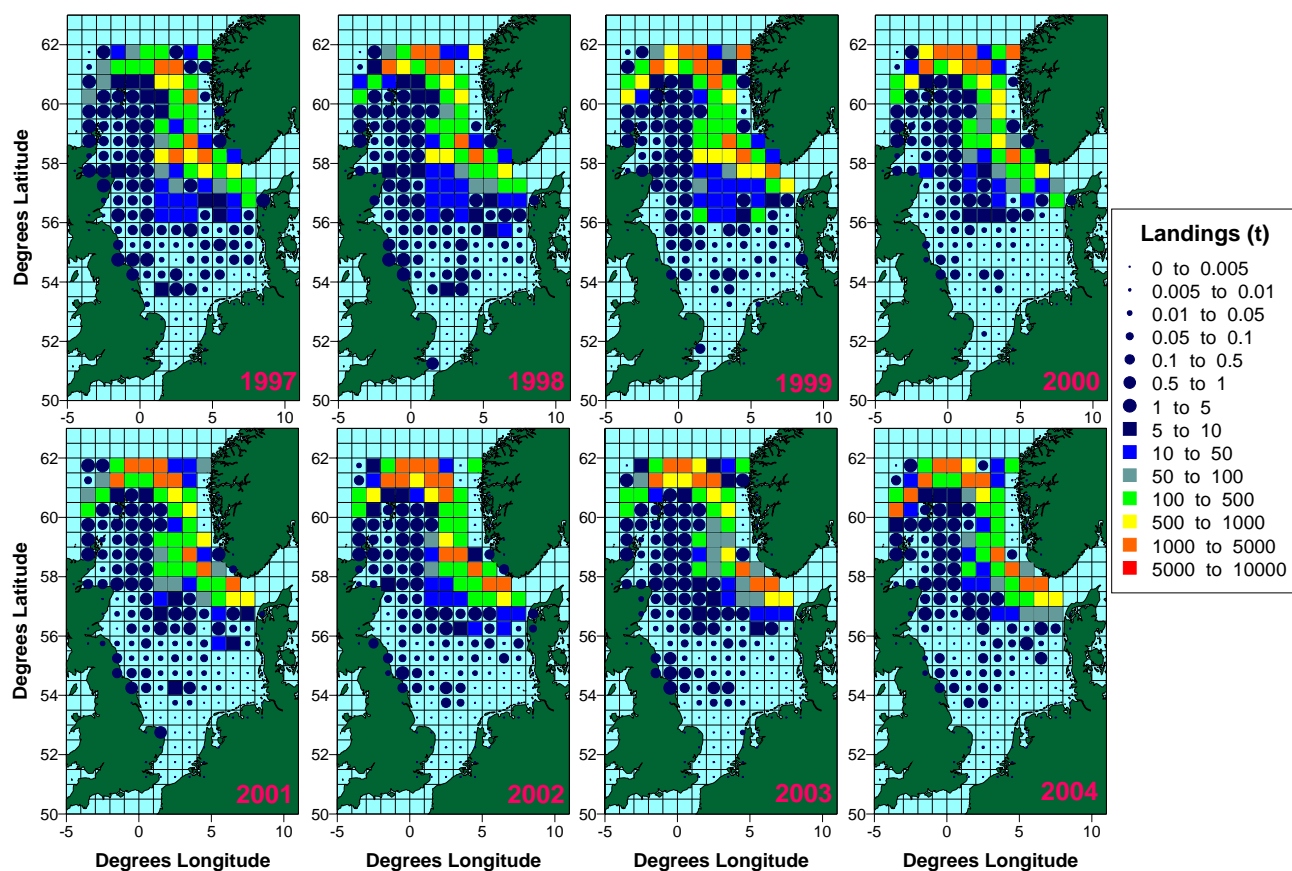


Figure 8.5.3.1.1.4. Spatial distributions of saithe landings across all fishing gears by countries not participating in the MAFCONS project in each of the years between 1997 and 2004.

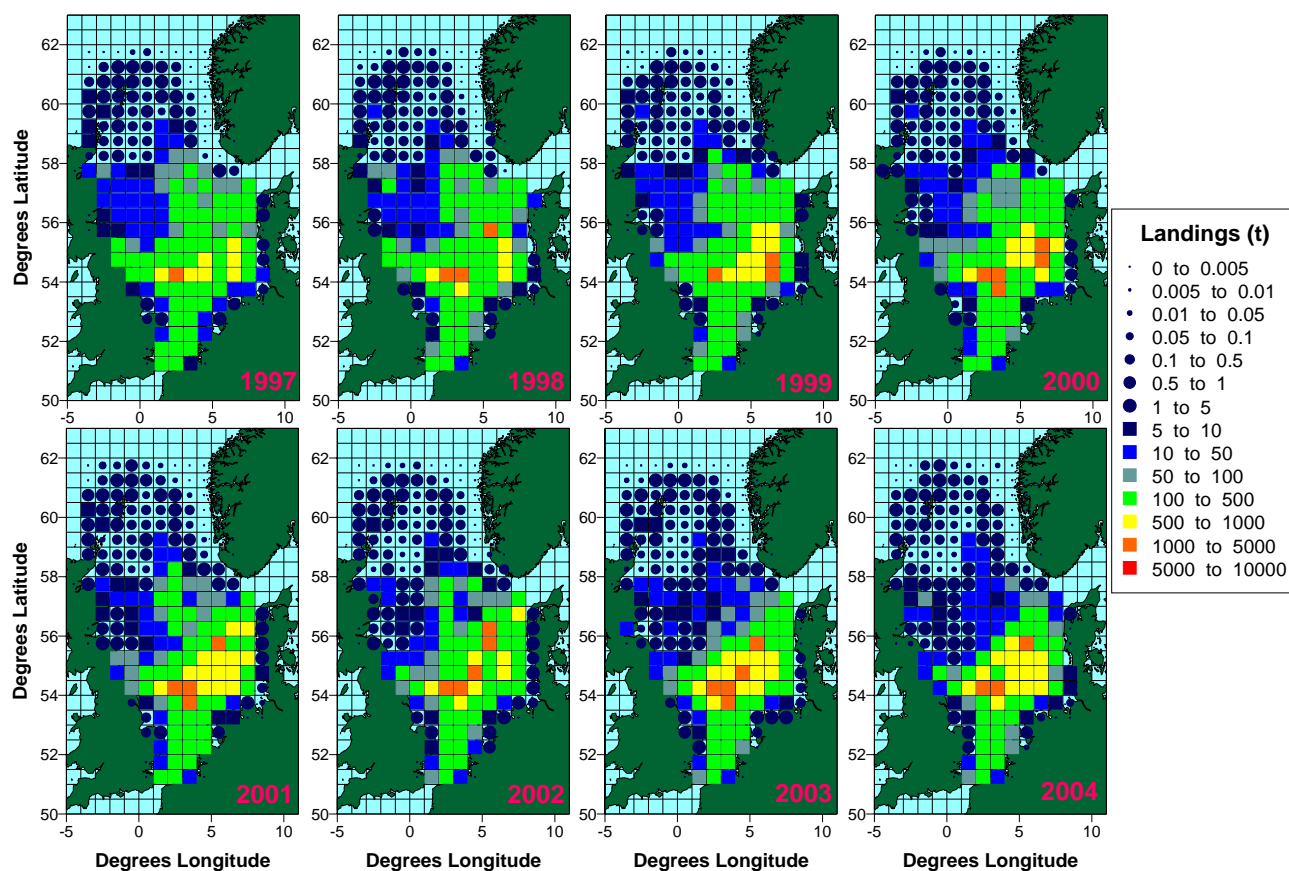


Figure 8.5.3.1.1.5. Spatial distributions of plaice landings across all fishing gears by countries not participating in the MAFCONS project in each of the years between 1997 and 2004.

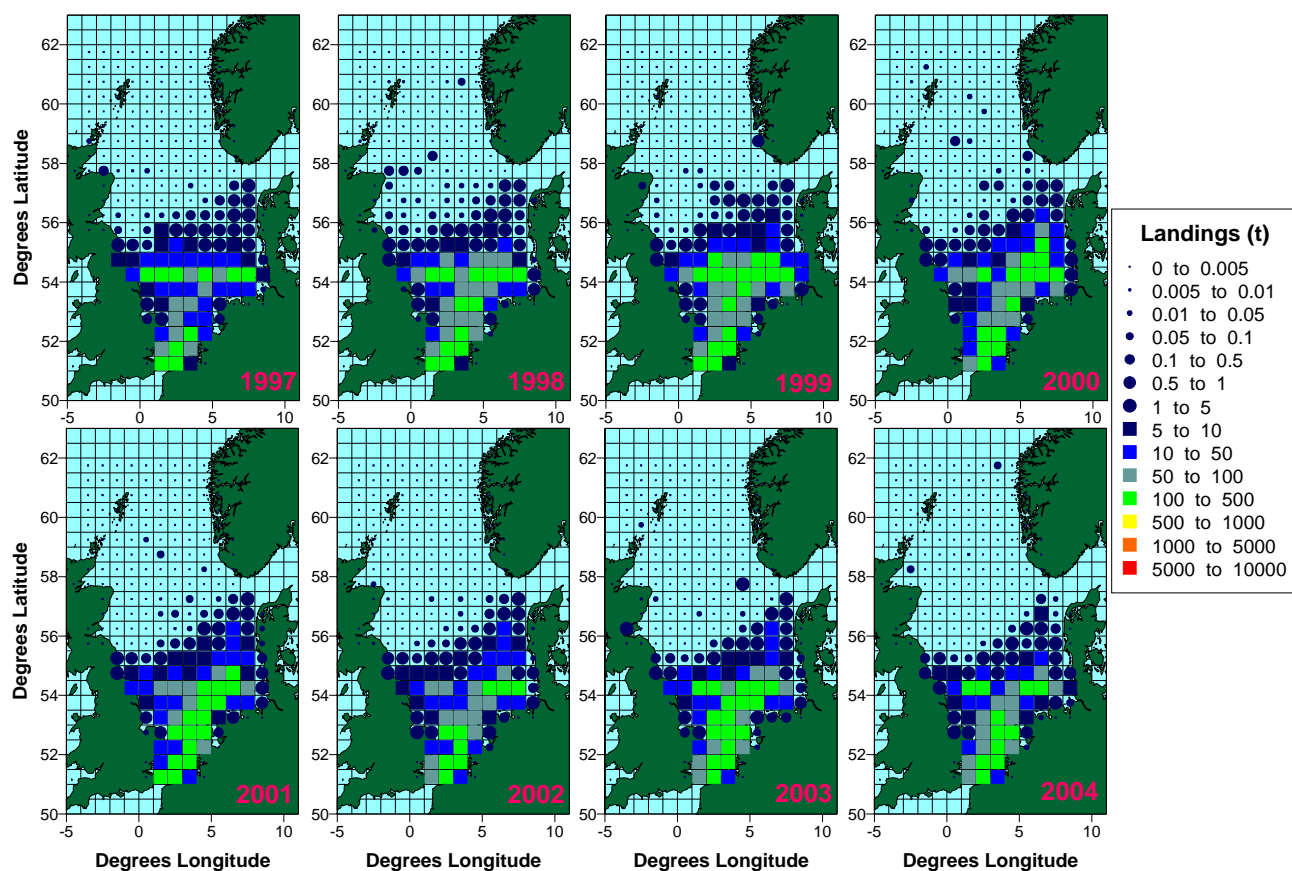


Figure 8.5.3.1.1.6. Spatial distributions of sole landings across all fishing gears by countries not participating in the MAFCONS project in each of the years between 1997 and 2004.

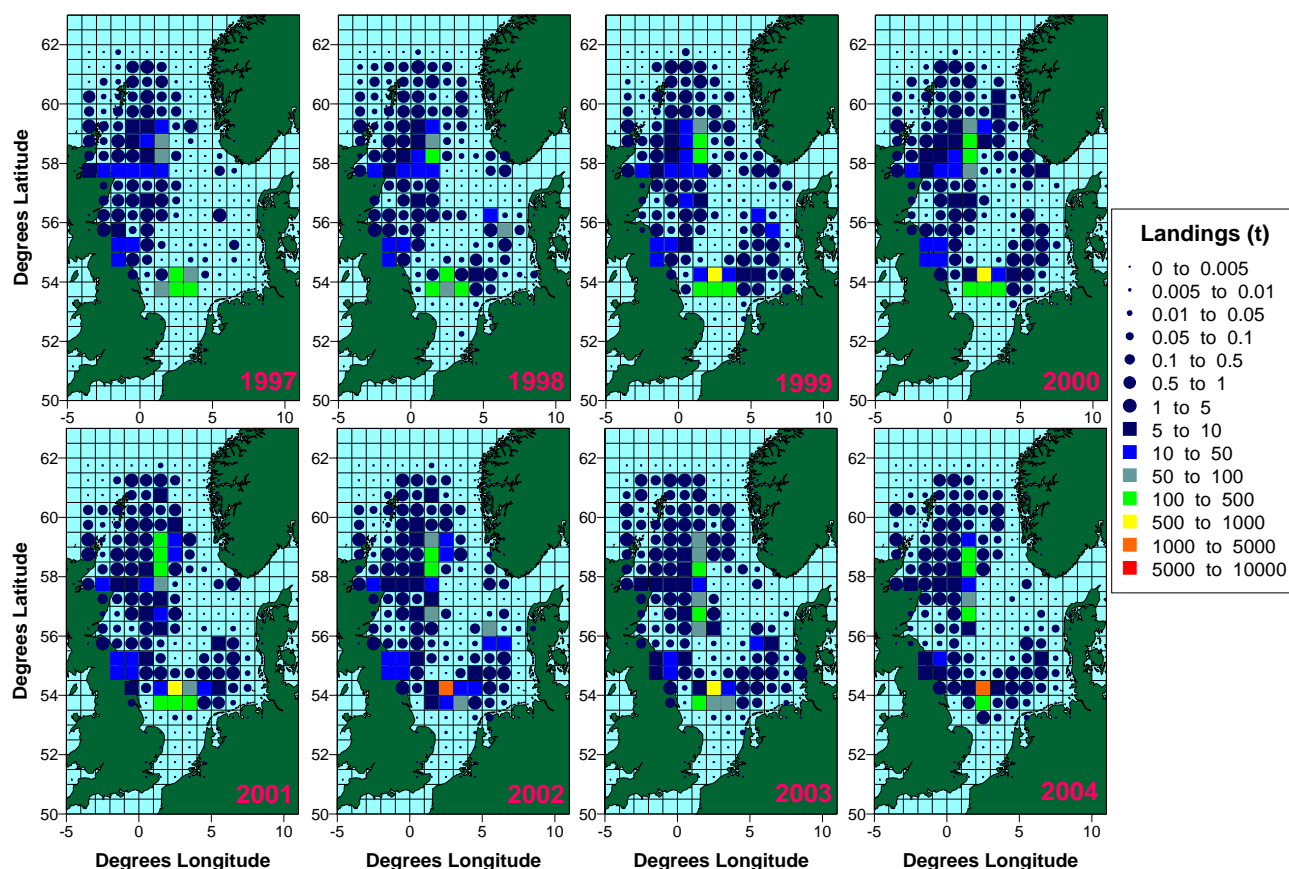


Figure 8.5.3.1.1.7. Spatial distributions of *Nephrops* landings across all fishing gears by countries not participating in the MAFCONS project in each of the years between 1997 and 2004.

8.5.3.1.2. Spatial trends in landings by gear category

Spatial patterns of modelled average annual landings of each species by the four main gear categories over the two time periods, 1997 to 2000 and 2001 to 2004, by the four non-MAFCONS countries (Figures 8.5.3.1.2.1 to 8.5.3.1.2.7) were again similar to those plotted for reported landings by the five MAFCONS countries (see Figures 8.3.2.2.1 to 8.3.2.2.7). Once again however, these spatial restrictions imposed on the modelled fleets were apparent, but again these seemed reasonable given the gears and the locations of the countries concerned.

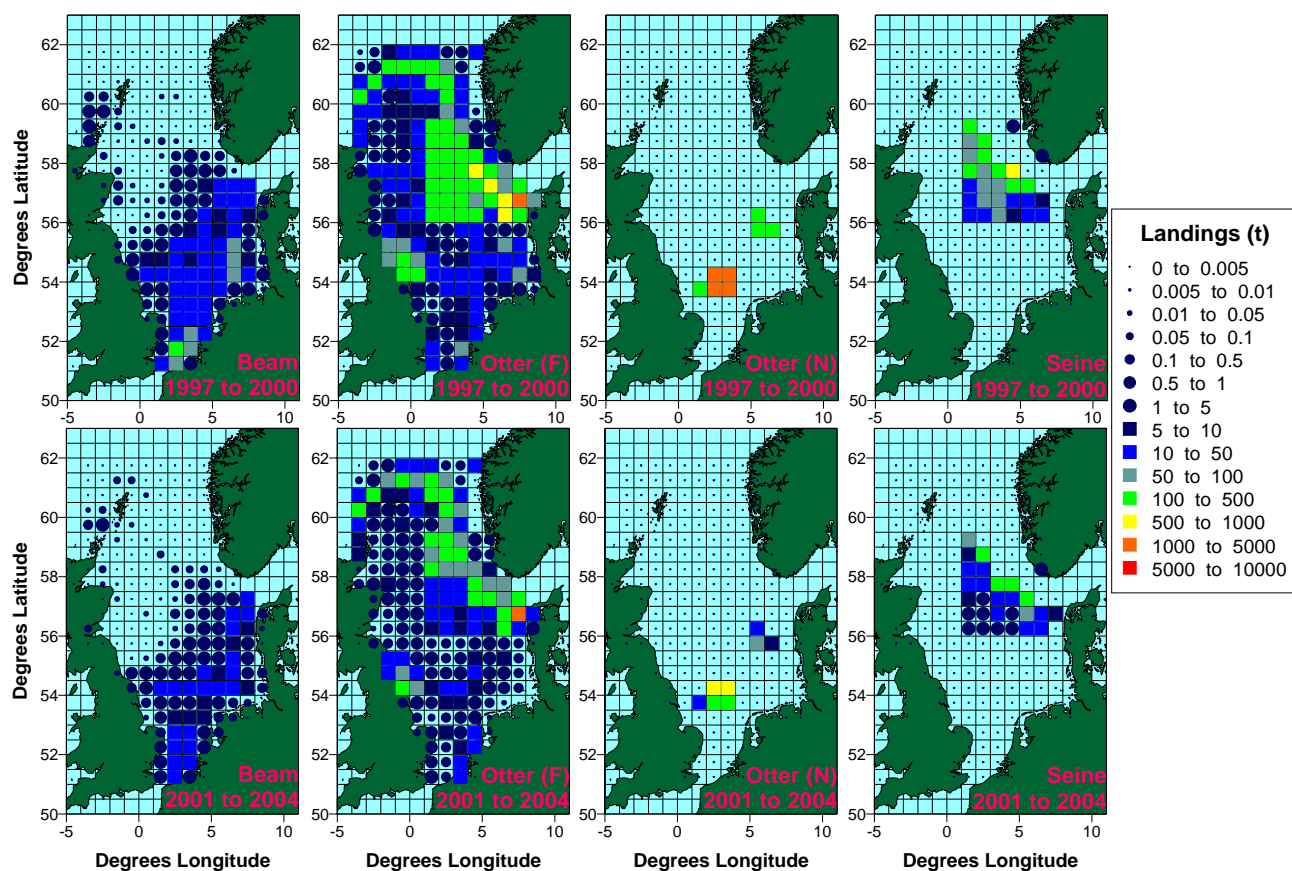


Figure 8.5.3.1.2.1. Spatial distributions in the average annual landings of cod taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by Belgium, Sweden, France, and Denmark in two four year periods, 1997 to 2000 and 2001 to 2004.

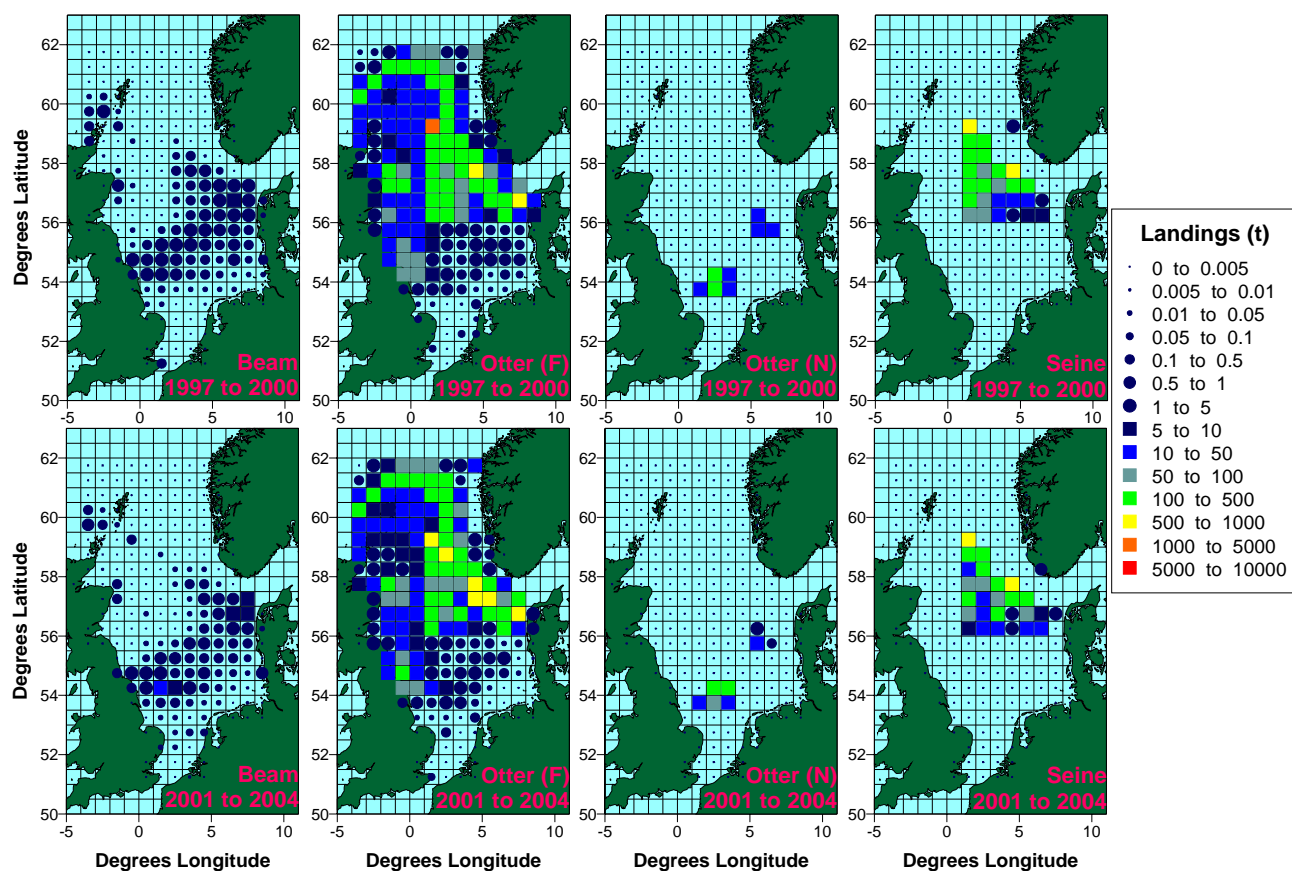


Figure 8.5.3.1.2.2. Spatial distributions in the average annual landings of haddock taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by Belgium, Sweden, France, and Denmark in two four year periods, 1997 to 2000 and 2001 to 2004.

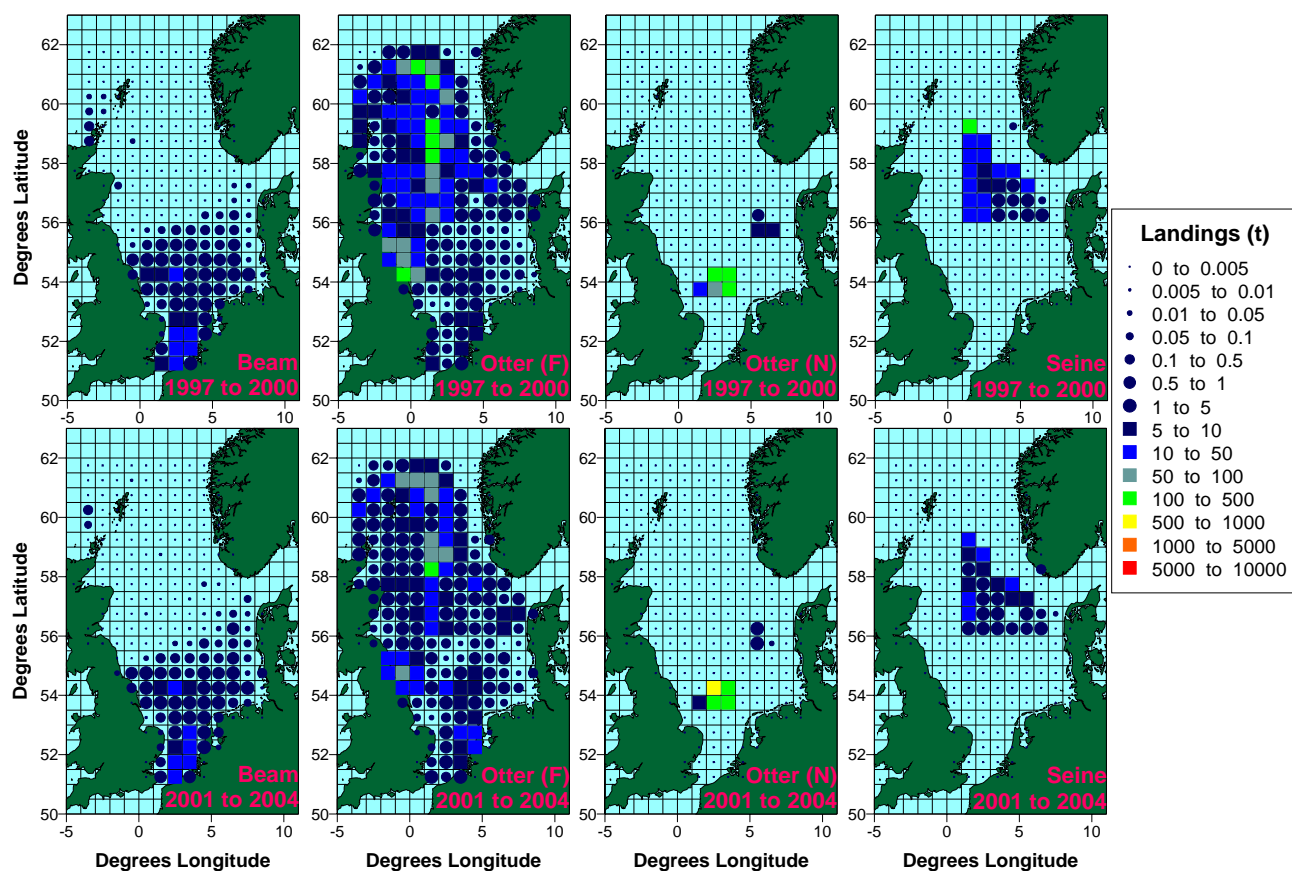


Figure 8.5.3.1.2.3. Spatial distributions in the average annual landings of whiting taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by Belgium, Sweden, France, and Denmark in two four year periods, 1997 to 2000 and 2001 to 2004.

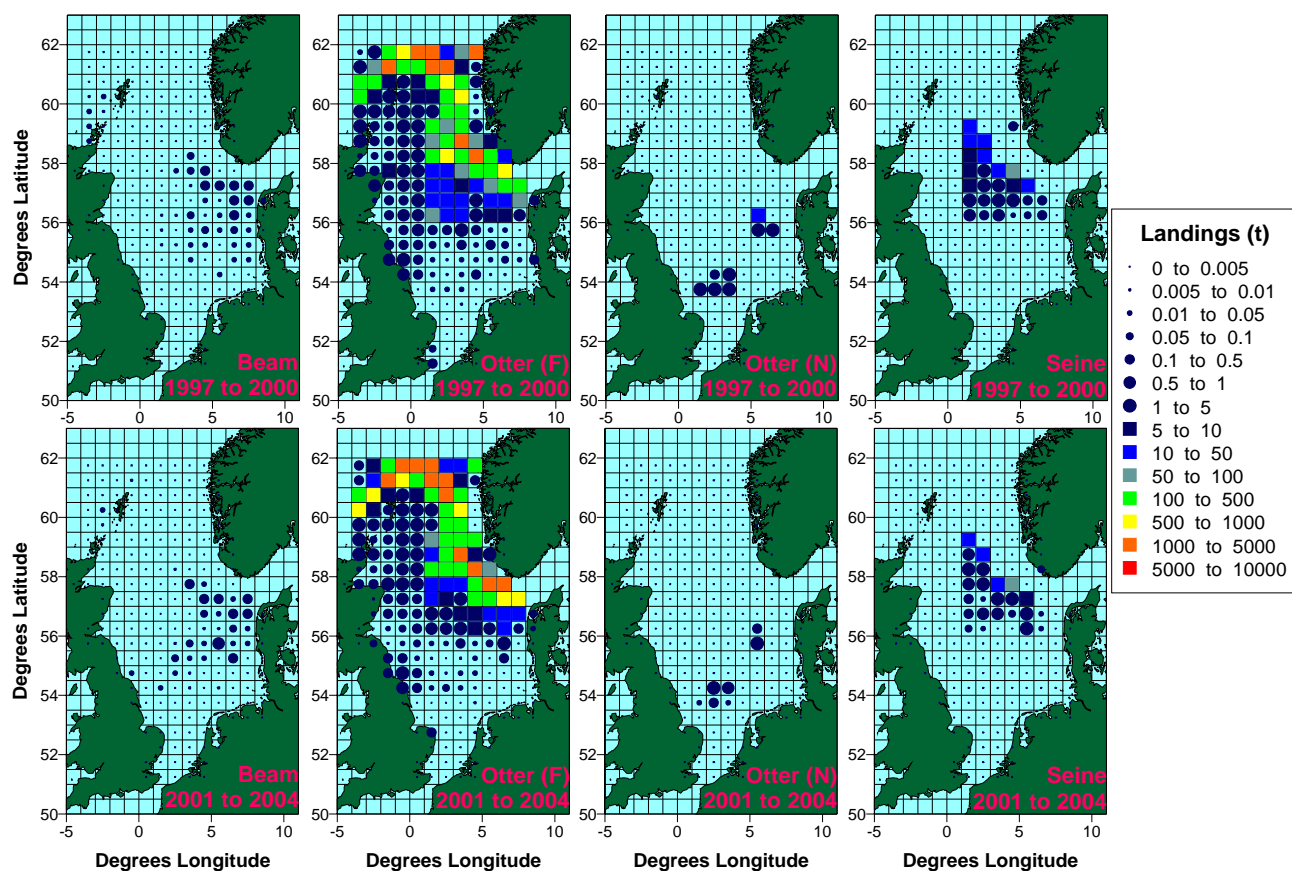


Figure 8.5.3.1.2.4. Spatial distributions in the average annual landings of saithe taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by Belgium, Sweden, France, and Denmark in two four year periods, 1997 to 2000 and 2001 to 2004.

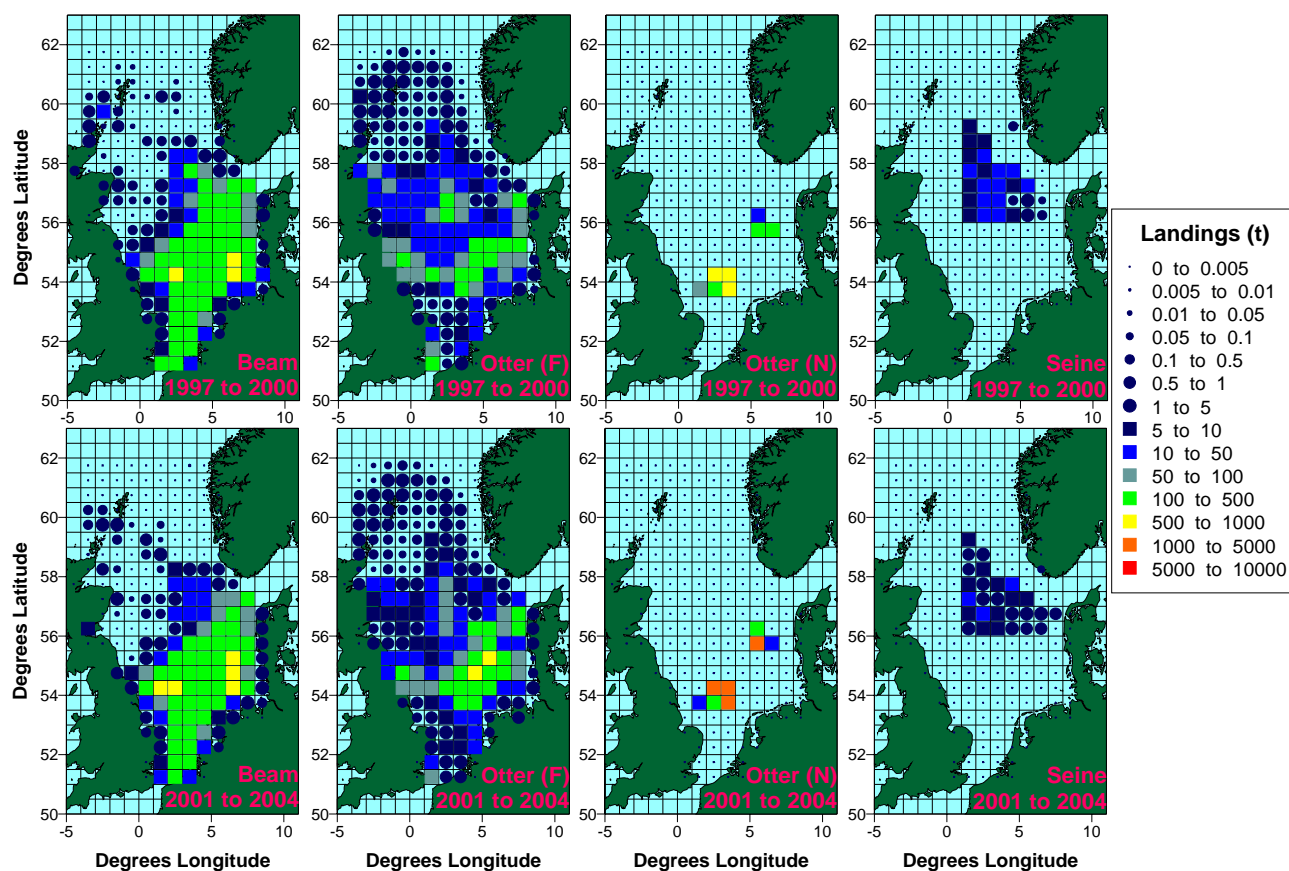


Figure 8.5.3.1.2.5. Spatial distributions in the average annual landings of plaice taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by Belgium, Sweden, France, and Denmark in two four year periods, 1997 to 2000 and 2001 to 2004.

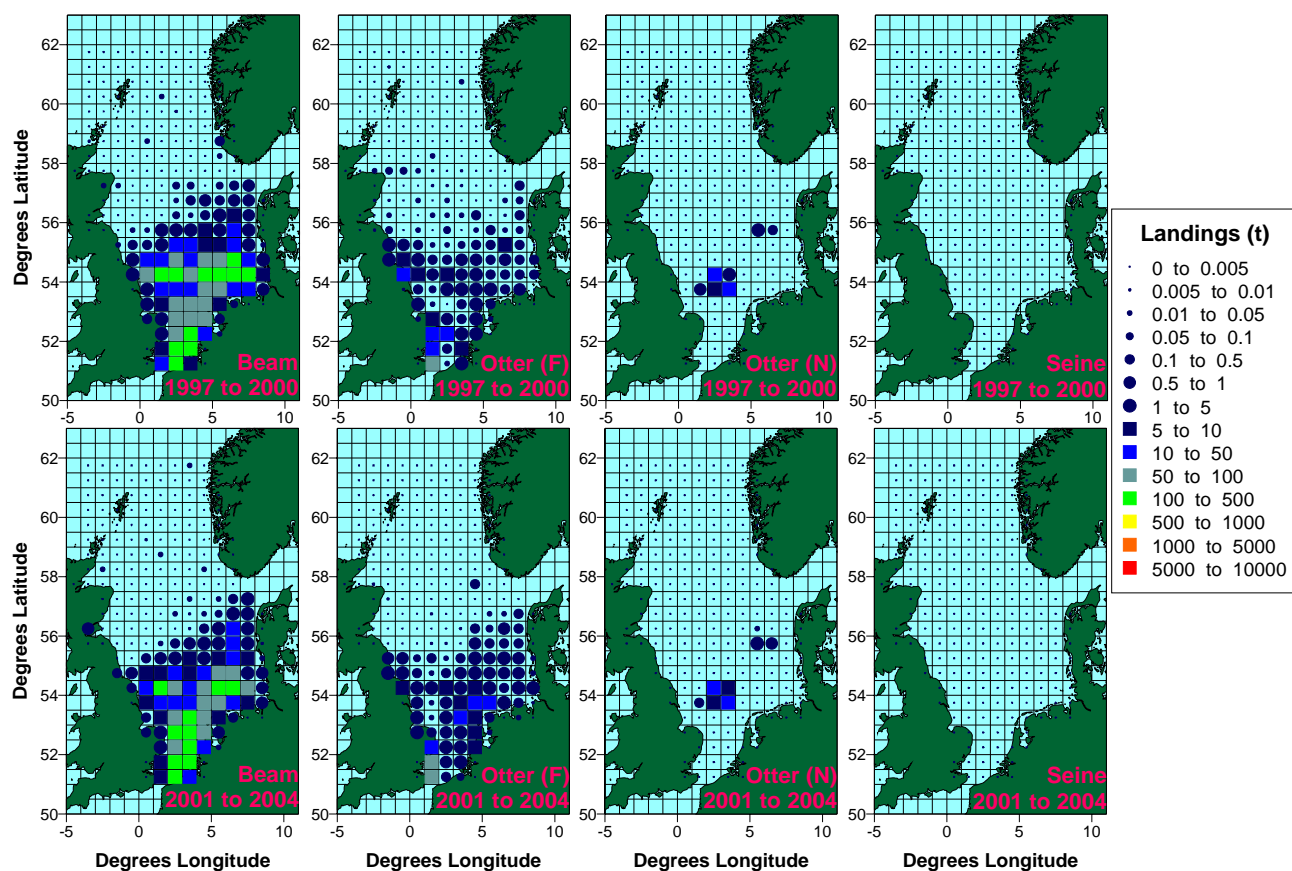


Figure 8.5.3.1.2.6. Spatial distributions in the average annual landings of sole taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by Belgium, Sweden, France, and Denmark in two four year periods, 1997 to 2000 and 2001 to 2004.

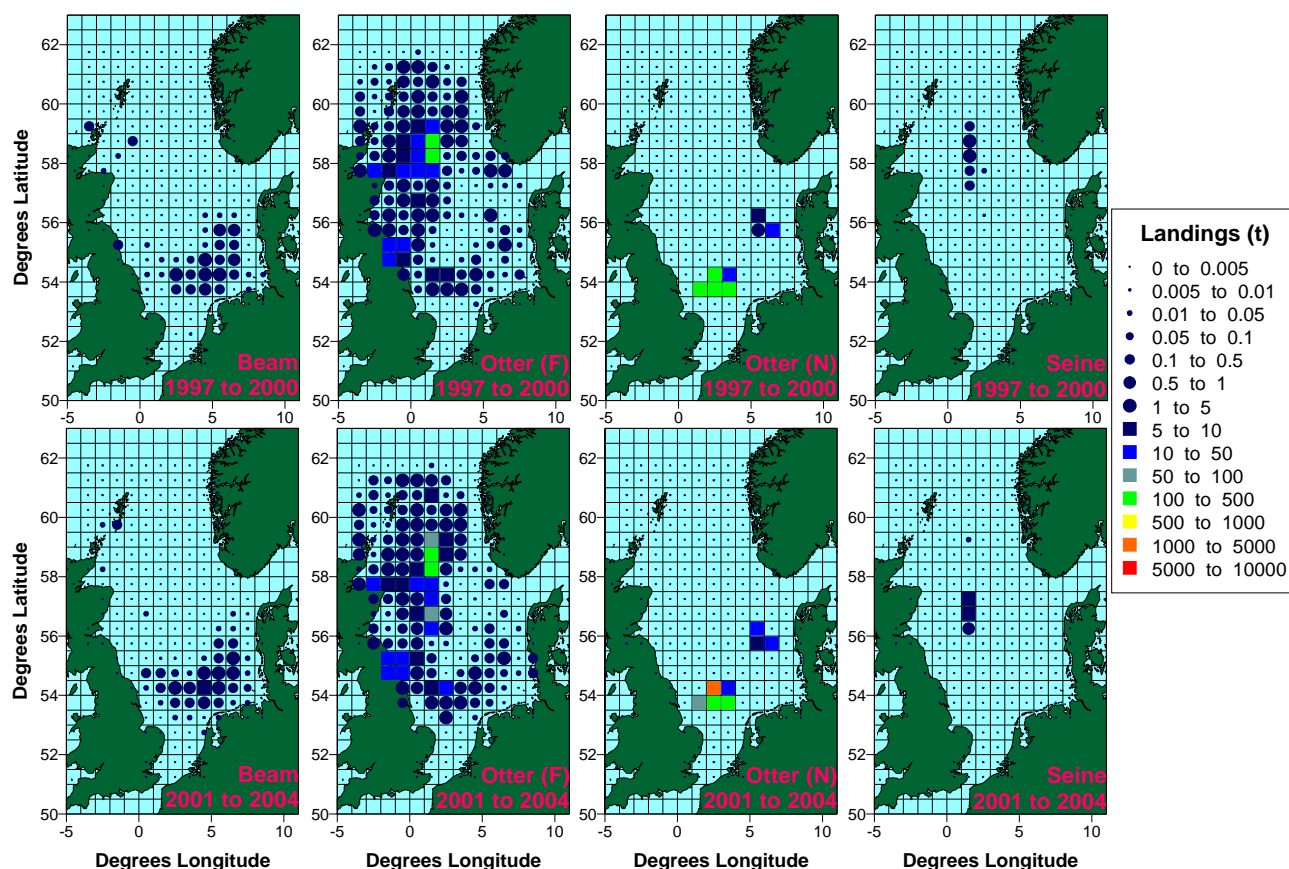


Figure 8.5.3.1.2.7. Spatial distributions in the average annual landings of *Nephrops* taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by Belgium, Sweden, France, and Denmark in two four year periods, 1997 to 2000 and 2001 to 2004.

8.5.3.1.3. Spatial trends in landings by country

The effects of spatial limitations imposed on the different fleet components on the distributions of modelled landings by each of the non-MAFCONS countries was particularly apparent when modelled average landings of each species by each country in each of the two time periods was plotted (Figures 8.5.3.1.3.1 to 8.5.3.1.3.7). The smoothest distribution was observed for Sweden, which only had one modelled fleet component, otter trawl targeting saithe, with no spatial restriction. The northern limits to the Belgian otter trawl fleet and the French beam trawl fleet, and the southern limit to the Danish beam trawl fleet, are clearly evident. Similarly, the western and southern limits to the Danish otter trawl and seine gear fleet components targeting cod are obvious, as are the restrictions to the Danish and French otter trawl fleets targeting saithe in the 37 northerly rectangles. Despite this though, the modelled landings distributions reflect the strong likelihood that each of the fleets would have preferentially operated in the most productive ICES rectangles that were closest to their home ports; a similar conclusion to the one drawn following examination of reported landings by the five MAFCONS countries (see Figures 8.3.2.3.1 to 8.3.2.3.7).

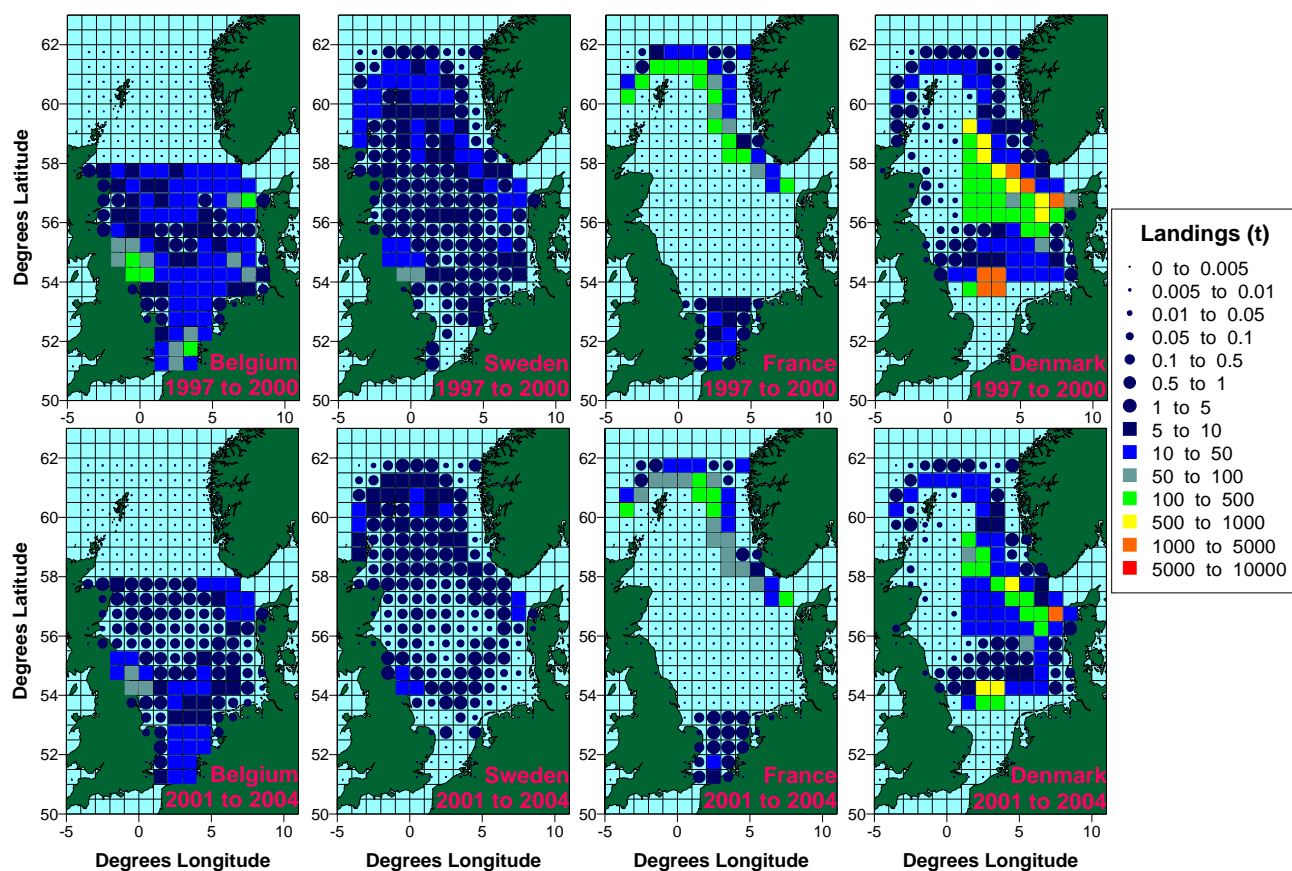


Figure 8.5.3.1.3.1. Spatial distributions of average annual landings of cod modelled for the four countries not contributing to the MAFCONS landings database, Belgium, Sweden, France and Denmark, in two four year periods, 1997 to 2000 and 2001 to 2004.

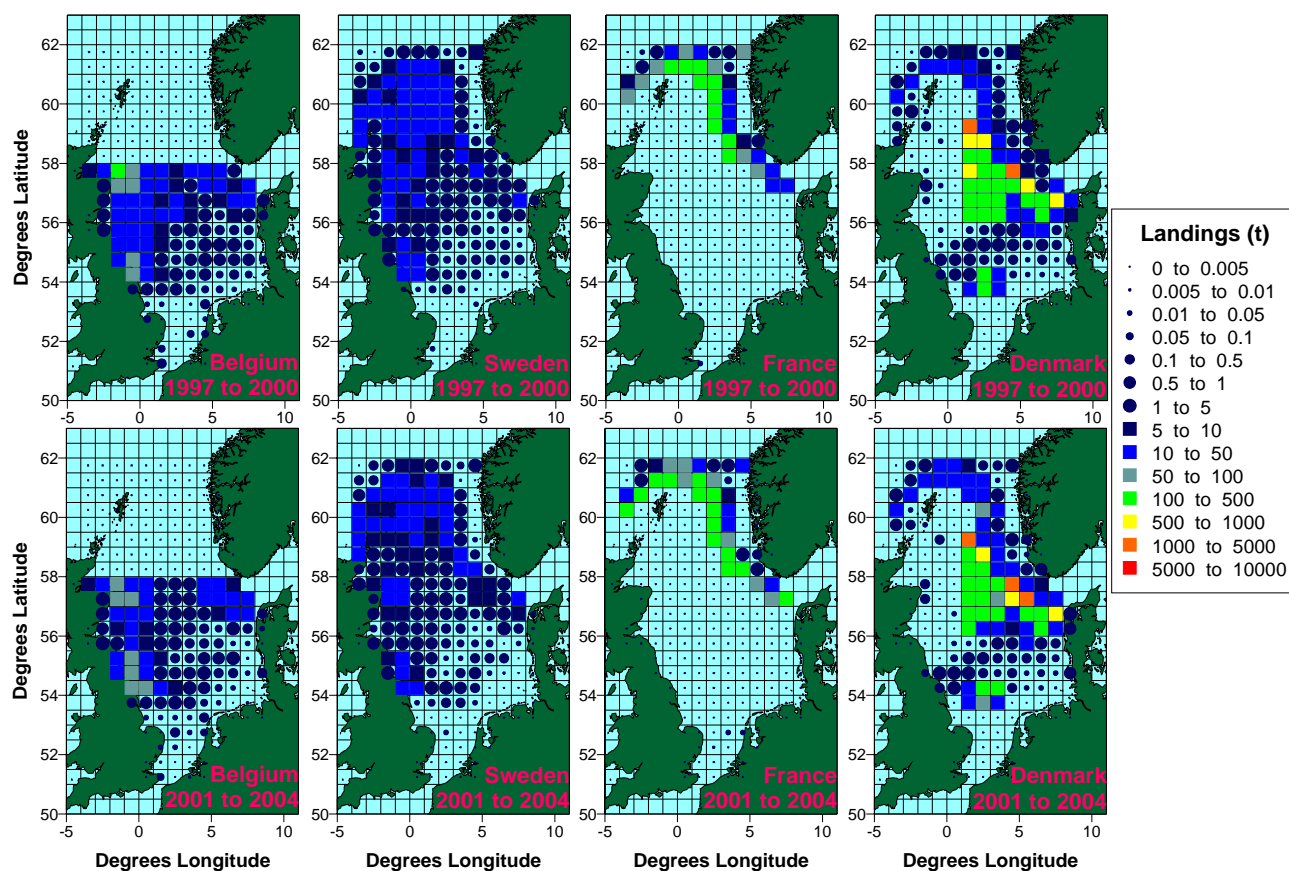


Figure 8.5.3.1.3.2. Spatial distributions of average annual landings of haddock modelled for the four countries not contributing to the MAFCONS landings database, Belgium, Sweden, France and Denmark, in two four year periods, 1997 to 2000 and 2001 to 2004.

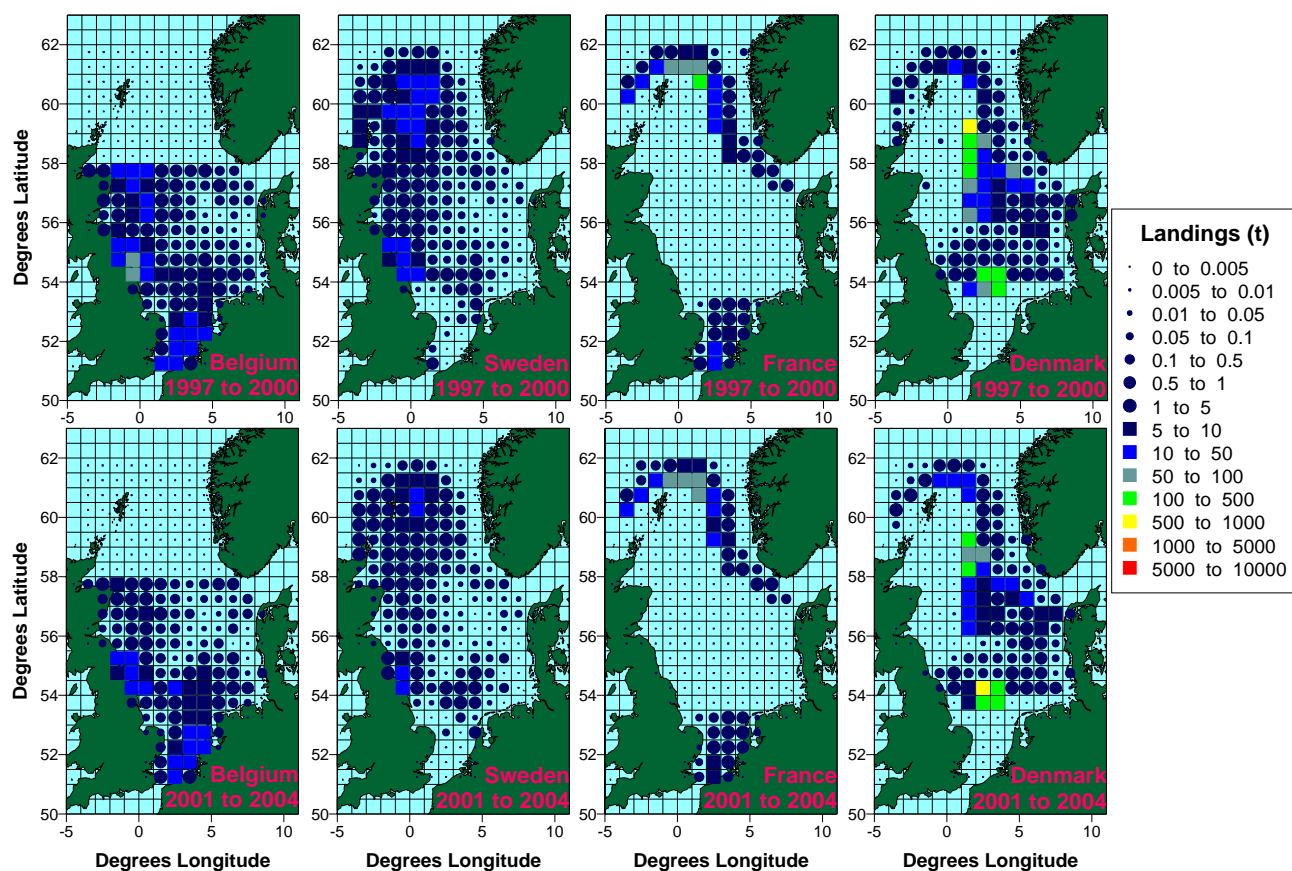


Figure 8.5.1.3.3. Spatial distributions of average annual landings of whiting modelled for the four countries not contributing to the MAFCONS landings database, Belgium, Sweden, France and Denmark, in two four year periods, 1997 to 2000 and 2001 to 2004.

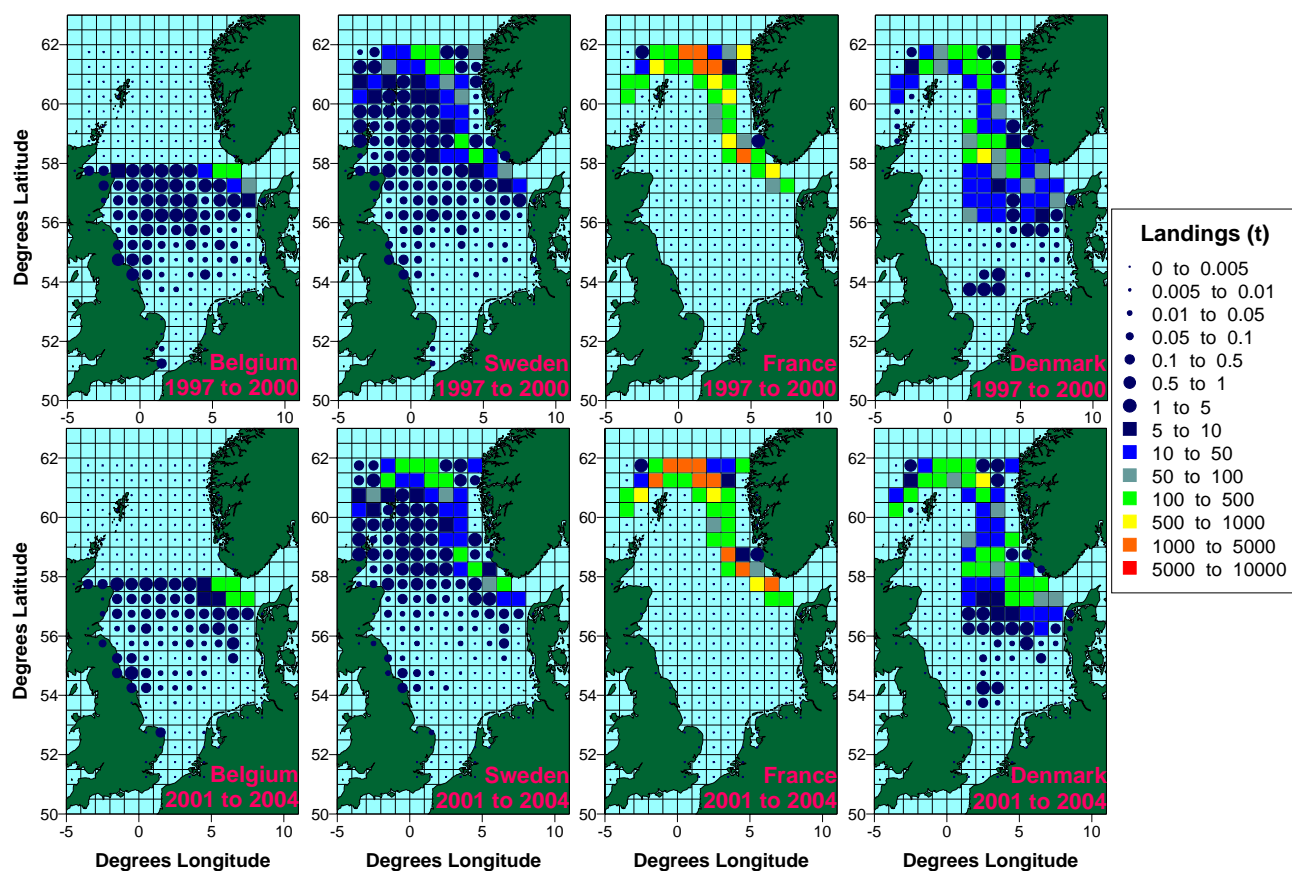


Figure 8.5.3.1.3.4. Spatial distributions of average annual landings of saithe modelled for the four countries not contributing to the MAFCONS landings database, Belgium, Sweden, France and Denmark, in two four year periods, 1997 to 2000 and 2001 to 2004.

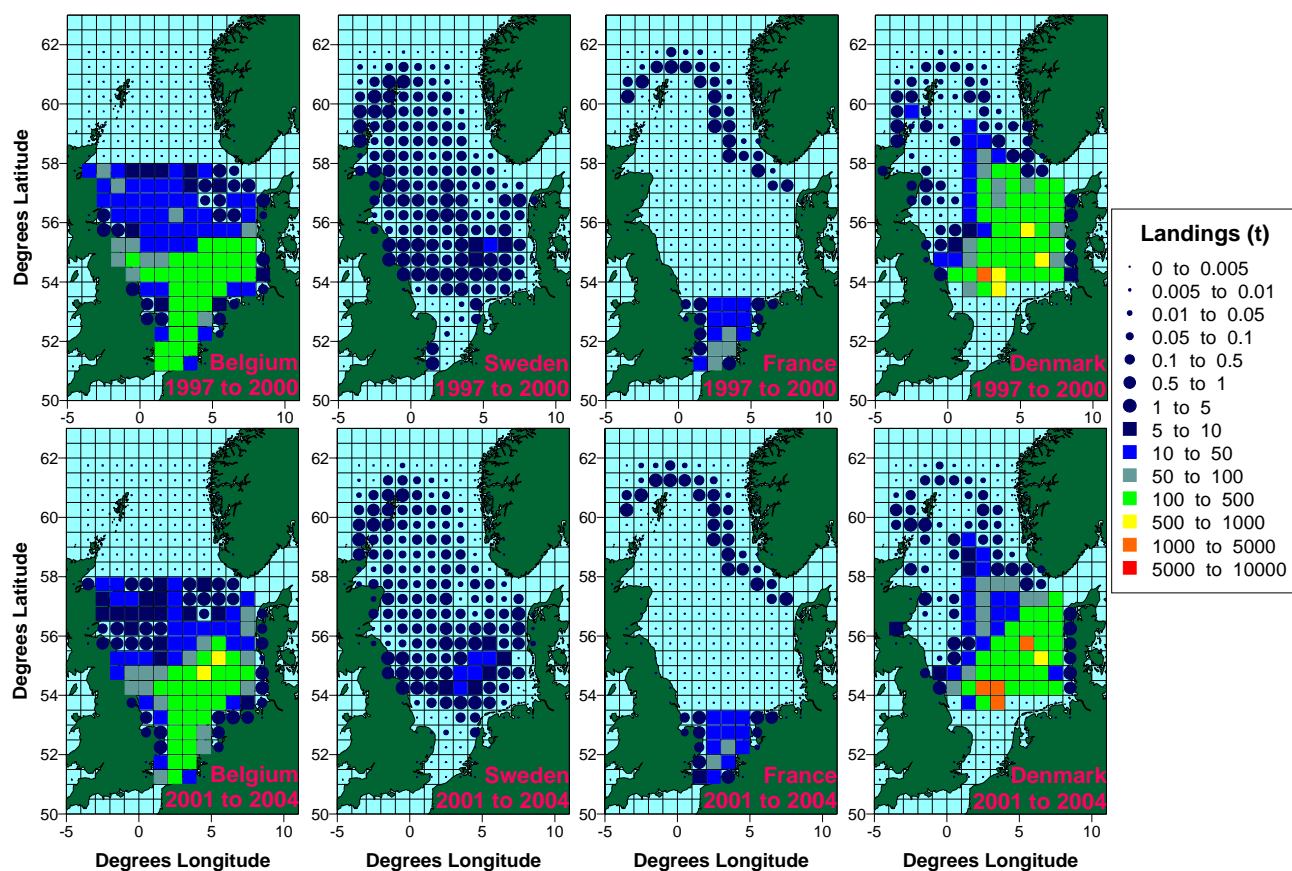


Figure 8.5.3.1.3.5. Spatial distributions of average annual landings of plaice modelled for the four countries not contributing to the MAFCONS landings database, Belgium, Sweden, France and Denmark, in two four year periods, 1997 to 2000 and 2001 to 2004.

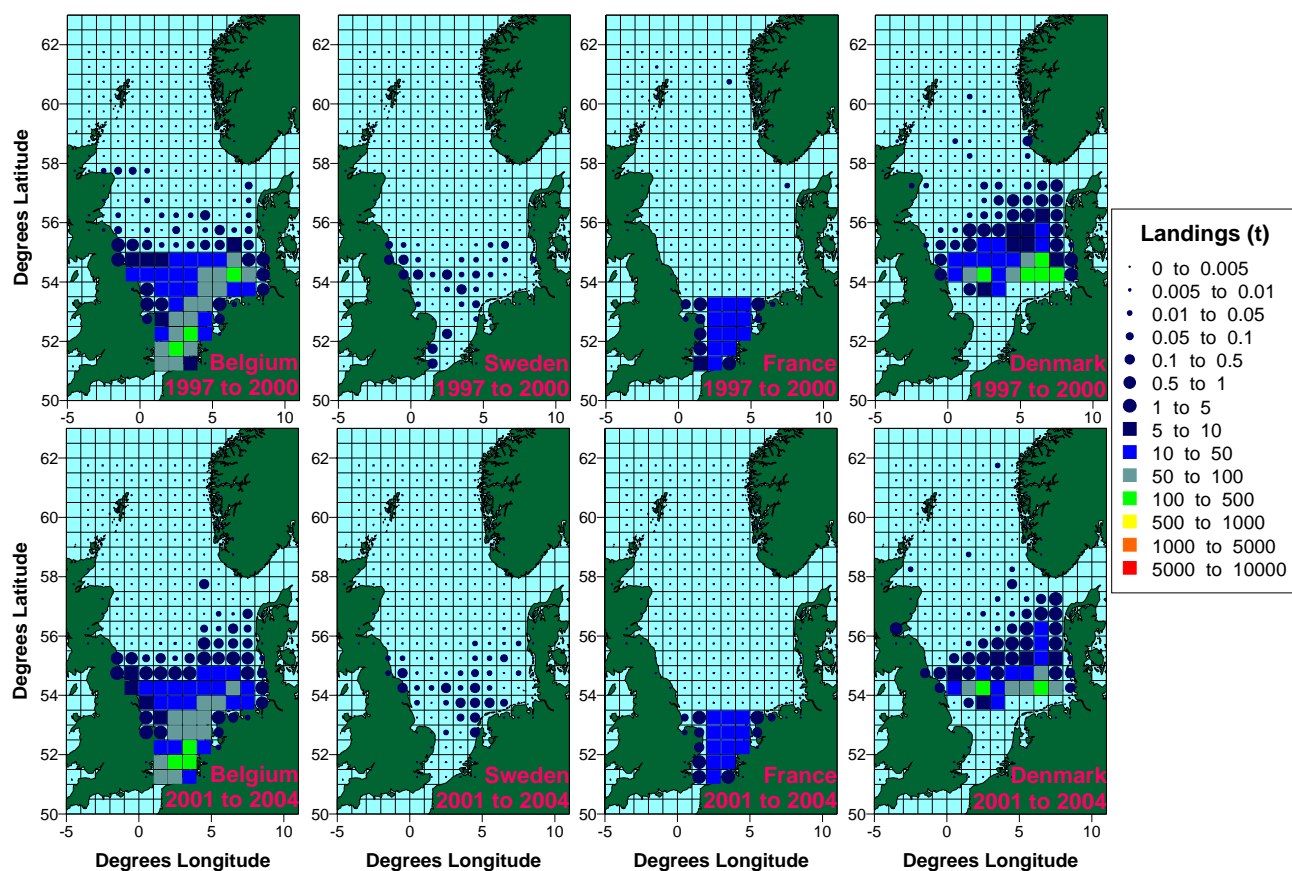


Figure 8.5.3.1.3.6. Spatial distributions of average annual landings of sole modelled for the four countries not contributing to the MAFCONS landings database, Belgium, Sweden, France and Denmark, in two four year periods, 1997 to 2000 and 2001 to 2004.

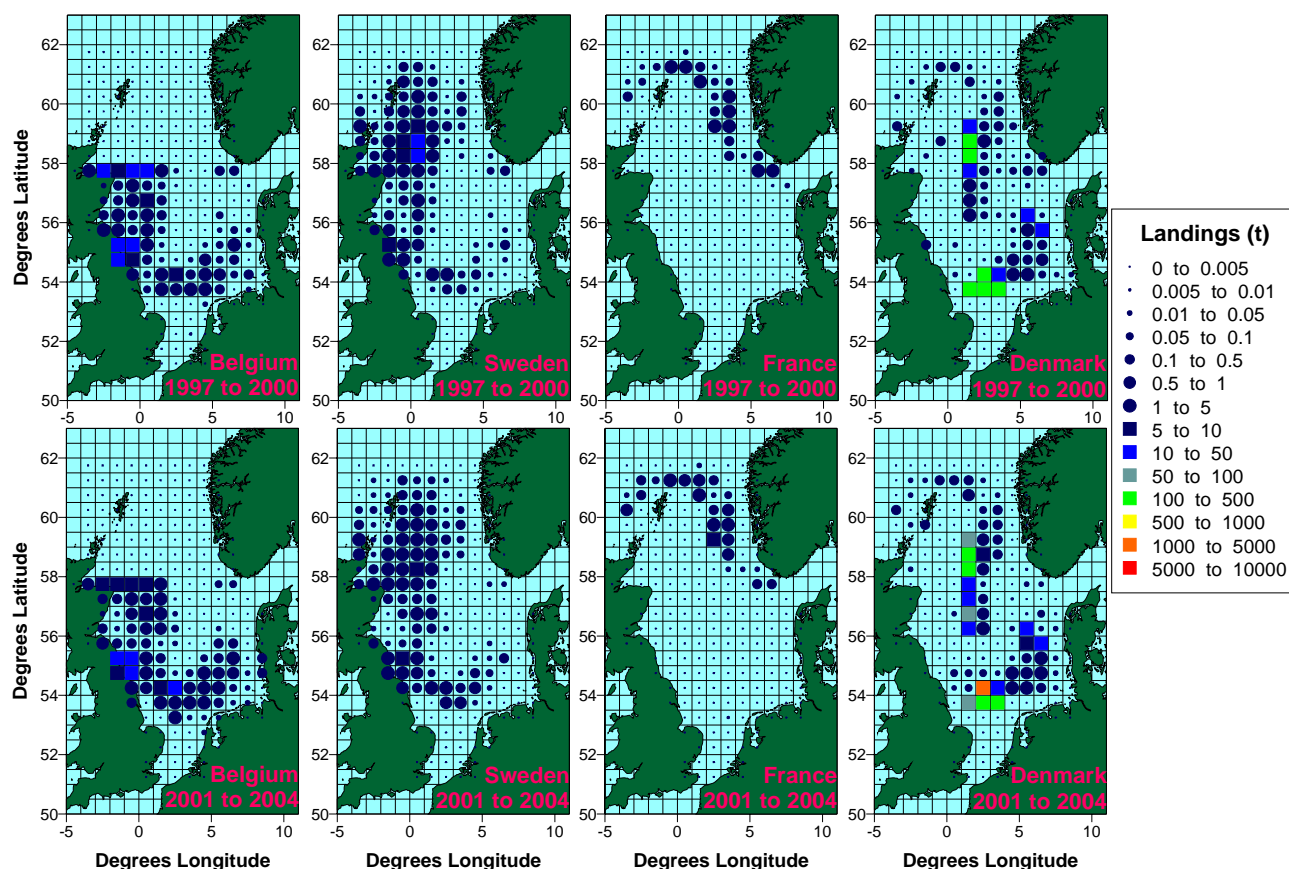


Figure 8.5.3.1.3.7. Spatial distributions of average annual landings of *Nephrops* modelled for the four countries not contributing to the MAFCONS landings database, Belgium, Sweden, France and Denmark, in two four year periods, 1997 to 2000 and 2001 to 2004.

8.5.3.2. Effort

No obvious temporal trend in modelled fishing effort was apparent for the four non-MAFCONS countries (Figure 8.5.3.2.1). As for the five MAFCONS countries for which effort data were available (see Figure 8.4.3.1), otter trawl was the most used gear, particularly directed at fish for human consumption (Figure 8.5.3.2.1). Beam trawl constituted between 25% and 33% of total effort by the non-MAFCONS countries (Figure 8.5.3.2.1), whereas for the five MAFCONS countries this proportion was slightly higher at around 40% (see Figure 8.4.3.1). However, the MAFCONS countries included the Netherlands, the European country with the largest beam trawl fleet. Seine gear use made up less than 10% of total effort (Figure 8.5.3.2.1), consisting entirely of Danish seiners, a similar situation to that of the MAFCONS countries where only one country, Scotland, had a significant seining fleet (see Figure 8.4.3.1). Of the four modelled non-MAFCONS countries, Danish fishing effort was clearly the most important (Figure 8.5.3.2.1).

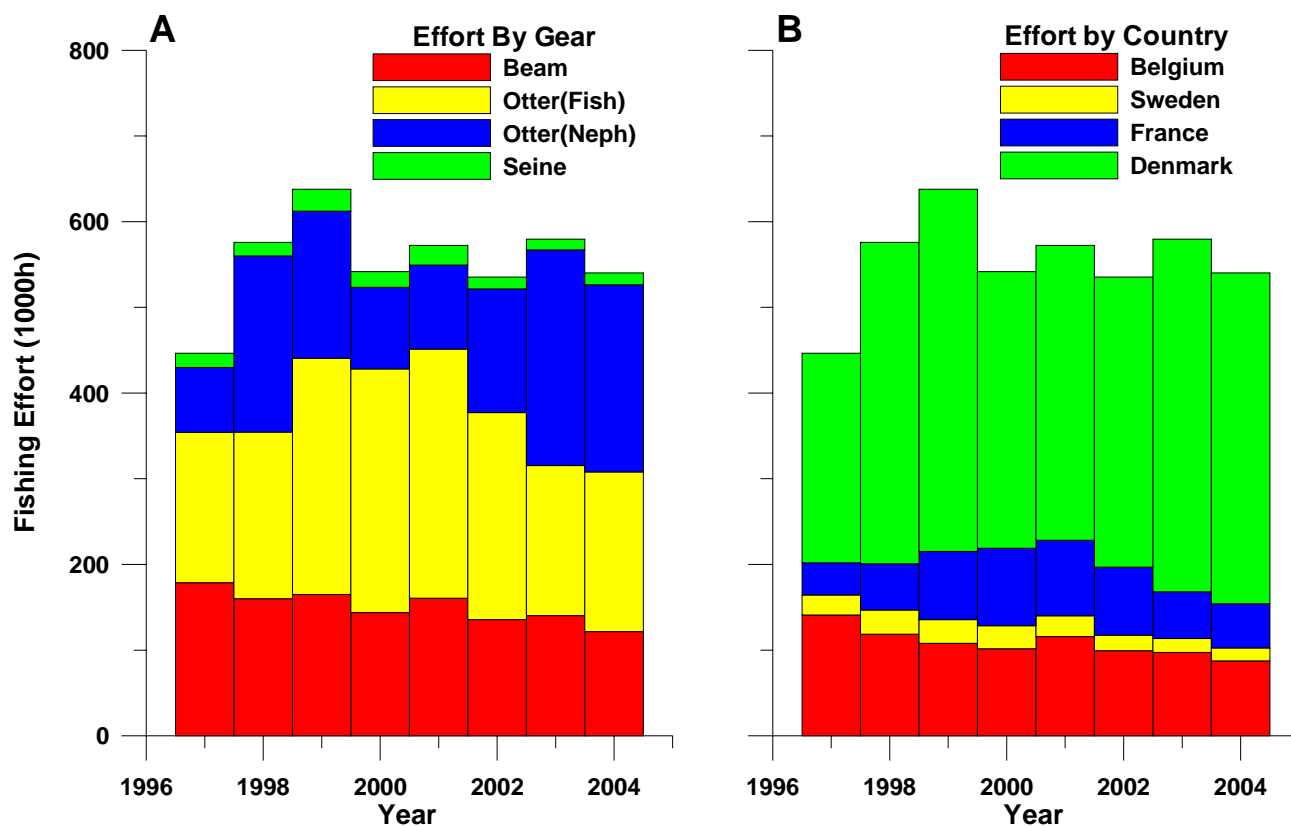


Figure 8.5.3.2.1. Trends in modelled annual fishing effort by the four countries not contributing to the MAFCONS database, Belgium, Sweden, France, and Denmark: (A) over the period 1997 to 2004 disaggregated by main gear category, and (B) over the period 1997 to 2004 disaggregated by the four countries.

Spatial distributions of effort by beam and otter trawl directed at fish for human consumption for the four non-MAFCONS countries combined (Figure 8.5.3.2.2) were similar to those plotted for the five MAFCONS countries (see Figure 8.4.4.1.1). Use of seine gear by the modelled non-MAFCONS countries had a more easterly distribution (Figure 8.5.3.2.2), reflecting the activities of Danish seiners, compared with the more westerly distribution of Scottish seine fishing (see Figure 8.4.4.1.1). Otter trawl directed at *Nephrops* by the modelled non-MAFCONS countries was restricted to the two grounds exploited by Danish trawlers in the central and eastern North Sea (ICES 2005; Figure 8.5.3.2.2), whereas otter trawl directed at *Nephrops* by the MAFCONS countries reflected activity of the Scottish *Nephrops* fleets operating in the Moray Firth, Firth of Forth and Fladden grounds (ICES 2005; see Figure 8.4.4.1.1). The spatial restrictions applied to each of the fleet components of the non-MAFCONS countries are clearly apparent in plots of the spatial distributions of total effort by each of the four countries (Figure 8.5.3.2.3).

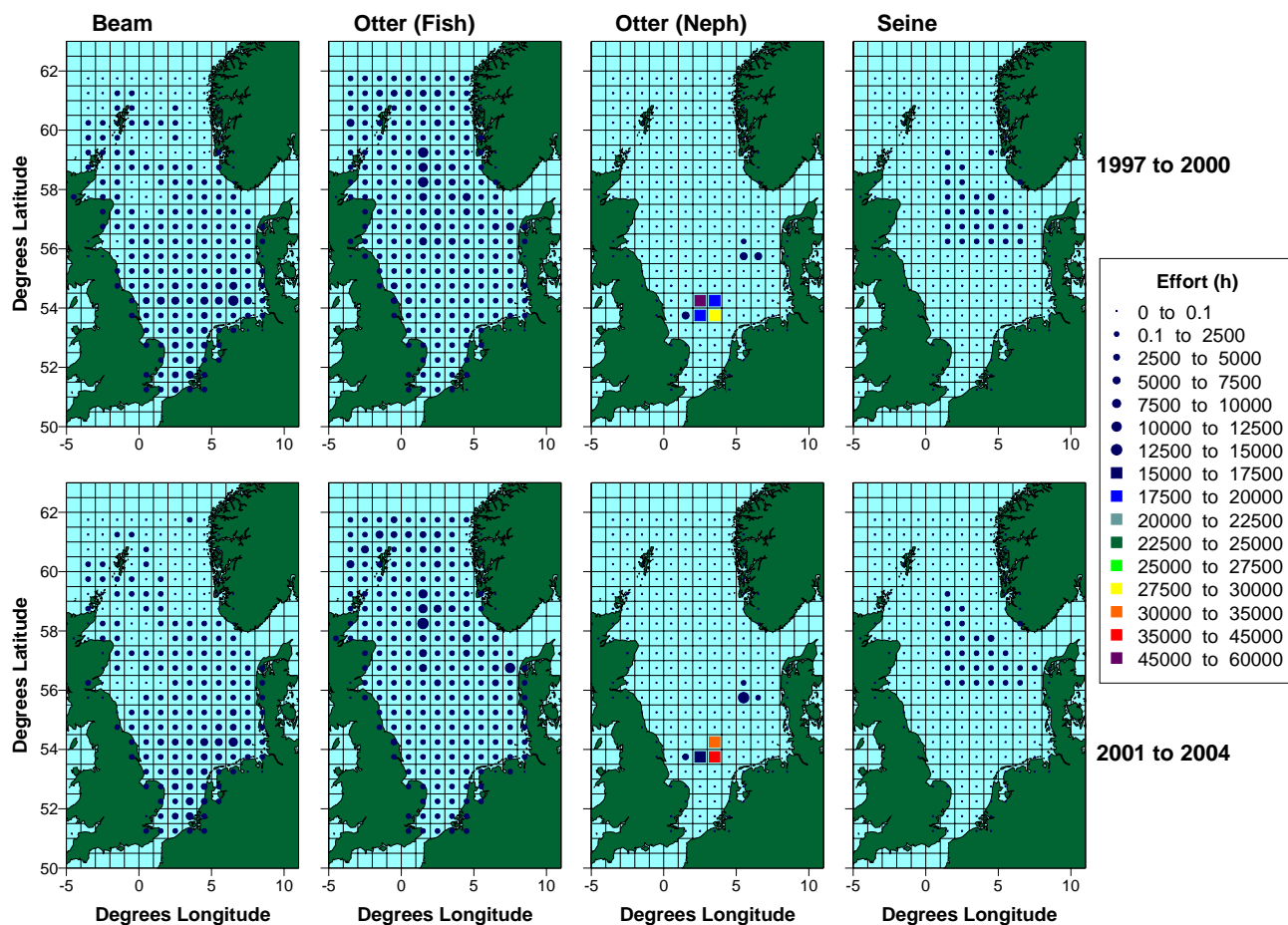


Figure 8.5.3.2.2. Distributions of modelled average annual fishing effort by beam trawl, otter trawl directed at fish for human consumption {Otter (Fish)}, otter trawl directed at *Nephrops* {Otter (Neph)}, and seine gear by the four countries not contributing to the MAFCONS database, Belgium, Sweden, France, and Denmark covering the periods 1997 to 2000, and 2001 to 2004.

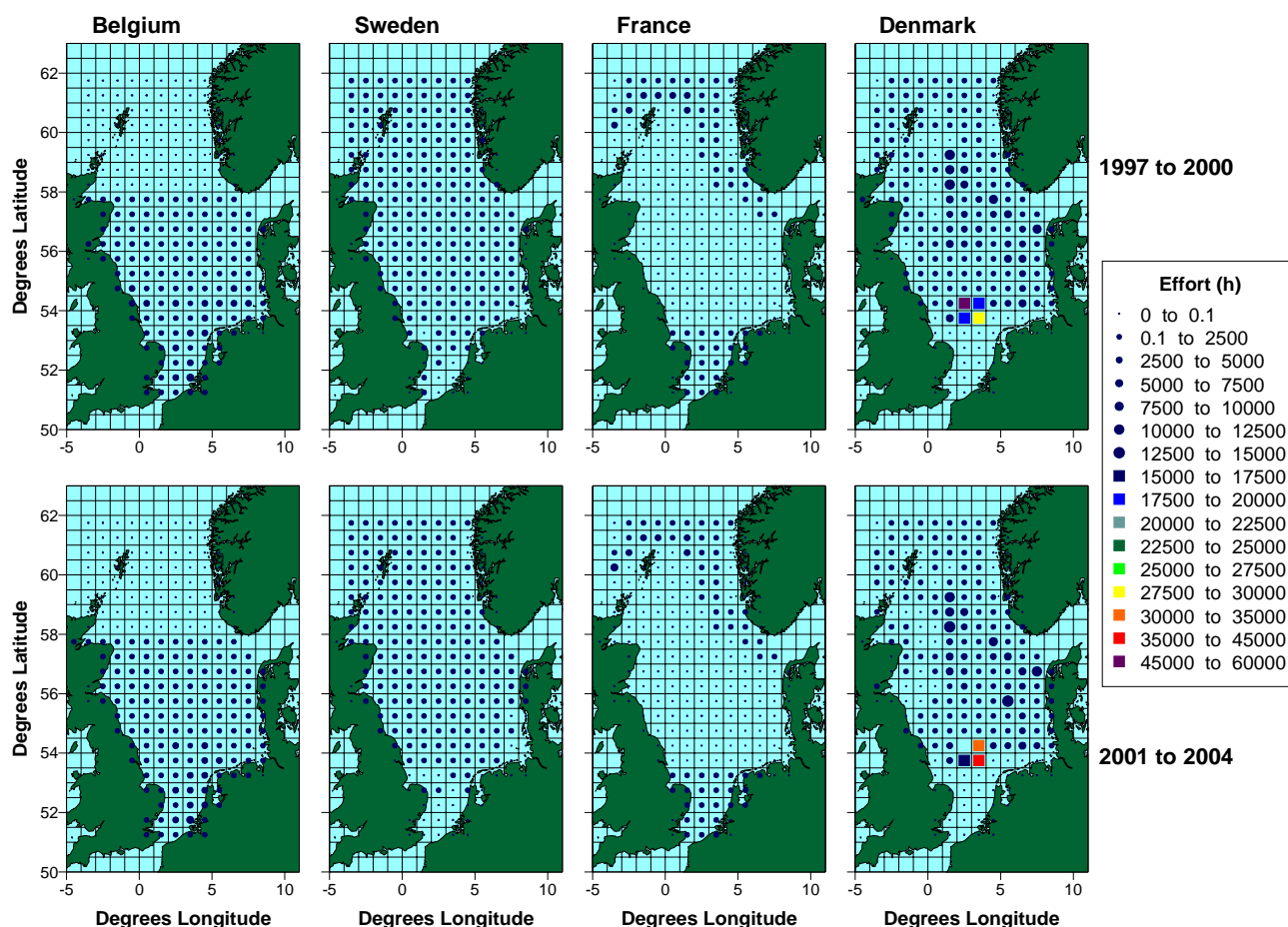


Figure 8.5.3.2.3. Distributions of modelled average annual fishing effort by vessels registered to Belgium, Sweden, France, and Denmark, in two periods, 1997 to 2000, and 2001 to 2004.

8.6. North Sea total international landings and fishing effort

8.6.1. Landings

8.6.1.1. *Annual totals*

Trends in total North Sea wide international landings reported by ICES (2005), disaggregated by country, were presented in Section 8.3.1 Figure 8.3.1.1. Prior to this, in Figure 8.3.1, we showed that landings data contributed by the MAFCONS participating countries, and incorporated into the MAFCONS landings database, more or less exactly matched the landings reported by ICES (2005) for these countries combined for five of the species examined. The exceptions were plaice, for which landings data in the MAFCONS database were consistently higher, and saithe, for which landings data in the MAFCONS database were consistently lower, than should have been the case, according to the ICES (2005) report. We also showed that the landings data supplied by the MAFCONS participating countries amounted to between 80% and 95% of total North Sea landings reported by ICES for haddock, whiting, plaice, sole, and Nephrops, and to between 55% and 65% of total landings for cod and saithe. The data reported by ICES therefore provide the best indication of

temporal variation in international landings of each species at the North Sea scale and there is no point in repeating these figures here.

In section 8.5.3.1 we then proceeded to model landings data for the Non-MAFCONS participating countries, essentially attempting to distribute the total landings reported by ICES (2005) for each country in each year across the ICES rectangles from which they were most likely to have been obtained. In a similar approach to that adopted for the five MAFCONS countries, we compared total modelled landings with total landings reported by ICES for the four countries combined (see Figure 8.5.3.1.1). This comparison suggested that our modelling approach reconciled modelled and reported total North Sea landings for cod, saithe, sole and Nephrops reasonably well, but that some problems were indicated for haddock, whiting and plaice. Our modelling approach actually models likely catches, not landings, and as such includes fish caught over quota and therefore discarded. The data reported by ICES (2005) that we have used in our analyses to date, and the data supplied by the MAFCONS participants for the landings database, are both landings data and therefore exclude discards. Thus failure to provide perfect fits, particularly for species that in our modelling approach were not specifically designated as “target species” for any of the modelled fleet components. It is well known that quota swapping between countries occurs. This may well reconcile some of the problems with these modelled data. Furthermore, that real difficulties exist in allocating landings to countries is evidenced by the fact that the ICES reports include an “unallocated” column that includes landings for each species that are not attributed to any country. Such unallocated landings are substantial for some species and some years, for example 7779t of cod in 1998, 5996t of haddock and 17,006t of saithe in 1997, over or close to 1000t of sole in every year, and 2045t of plaice in 1999 (ICES 2005). The issue of unallocated landings affects both the modelled data for the non-MAFCONS participating countries as well as the reported data for the MAFCONS countries.

Thus while repetition of the temporal trends in total landings is not necessary here, it is certainly worth examining how well the total combined reported (for the five MAFCONS countries) and modelled (for the four non-MAFCONS countries) landings data compare with the total landings data reported for ICES. Figure 8.6.1.1.1 compares MAFCONS total landings for the seven species in each year with the totals reported by ICES for the nine countries combined. In all cases the main “signal” in the trends was captured by the MAFCONS data, in that significant fits to the data were obtained with slopes very close to one. However, landings of haddock and plaice in the MAFCONS data base were consistently higher than suggested by the ICES data for these countries, while MAFCONS saithe landings data appeared consistently lower. Total haddock discarding rates as a percentage of total landings for human consumption each year varied between 36 and 288% (ICES 2005). Thus raising landings to catch in the human consumption fishery required multipliers of between 1.36 and 3.88, with an average multiplier of 1.93 over the eight years. Applying one minus the reciprocal of this value (ie $1 - 1/1.93 = 0.48$) to the haddock catches modelled for the four non MAFCONS countries, ie essentially applying an average annual discard rate to these modelled data, suggests that on average 8,500t of haddock would have been discarded annually by these four countries. Such a discard rate accounts for almost all the haddock discrepancy observed in Figure 8.6.1.1.1, implying that levels of discarding not taken into account in modelling the data for the non-MAFCONS countries could easily explain the observed difference. Data for plaice discards suggest that over the period 1997 to 2004, the discarded weight of plaice was equal to between 18% and 145% of the landed weight, implying landings weight to catch weight multipliers of 1.18 to 2.45, with an average multiplier of 1.85 over the eight years. If one minus the reciprocal of this value is again the “average plaice discard rate”, ie 0.46, then discards from the modelled plaice catches for

the four non-MAFCONS countries would have amounted to between 10,000t and 15,000t, sufficient to explain most of the discrepancy observed in Figure 8.6.1.1.1. Finally high levels of unallocated saithe landings (ICES 2005) explain why the reported and modelled landings by the nine main fishing nations operating in the North Sea fail to sum to the total North Sea landings reported. Finally Figure 8.6.1.1.2 examines annual variation in the ratio ACFM total landings divided by MAFCONS total landings. This figure suggests that the MAFCONS landings data, particularly considering the provisos discussed above, reflect trends in the ACFM total well.

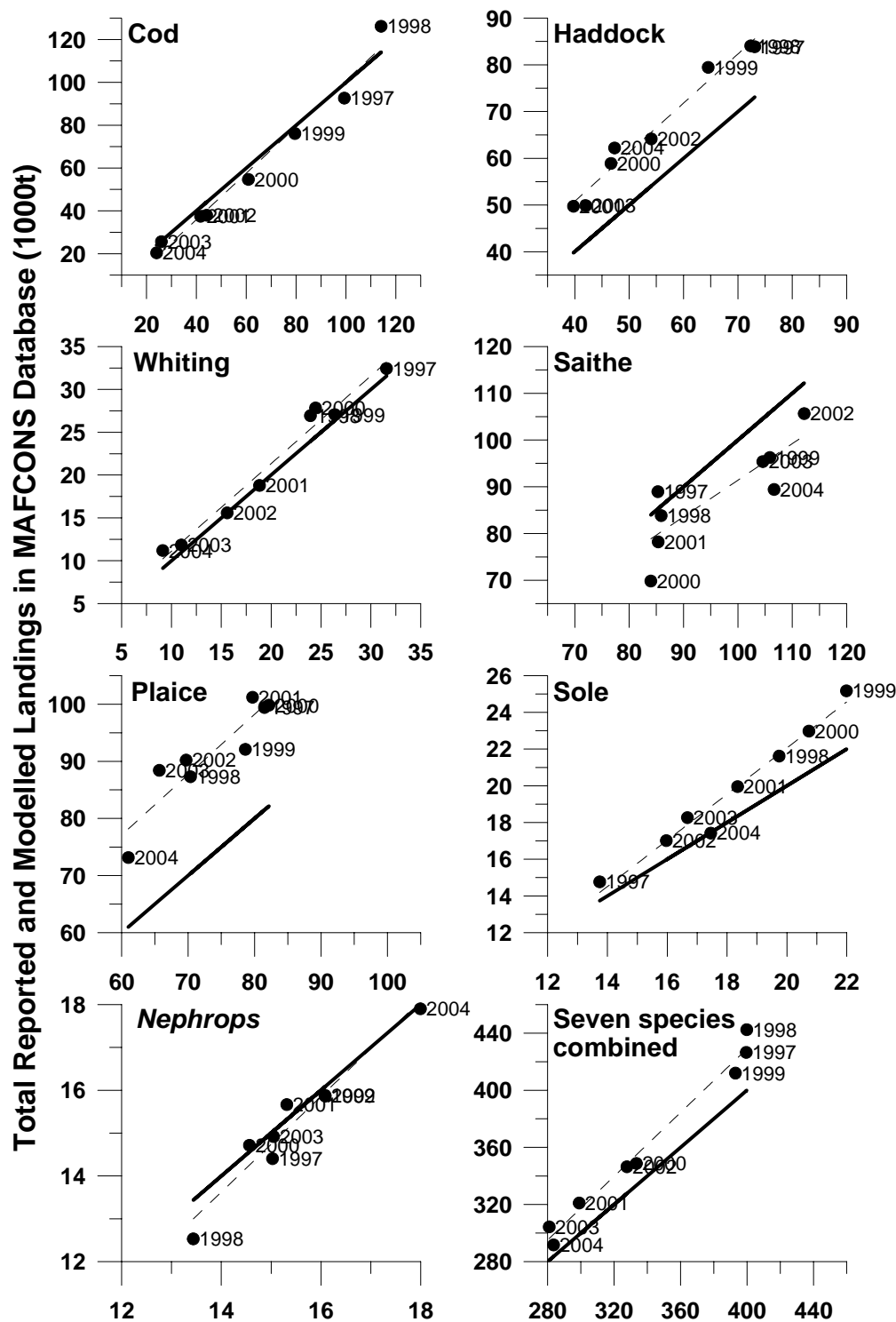


Figure 8.6.1.1.1. Correspondence between the landings data analysed to determine spatial patterns in landings as part of the MAFCONS project and the data reported to ICES for the stock assessments. Plots show total annual landings of each species, and all seven species combined, derived from the ACFM report (ICES 2005) summarising the ICES advice following the 2005 stock assessment process and annual summed reported and modelled landings data for the nine countries with major fishing operations in the North Sea. Solid lines indicate relationship expected for perfect agreement and dashed lines show linear fits to the data.

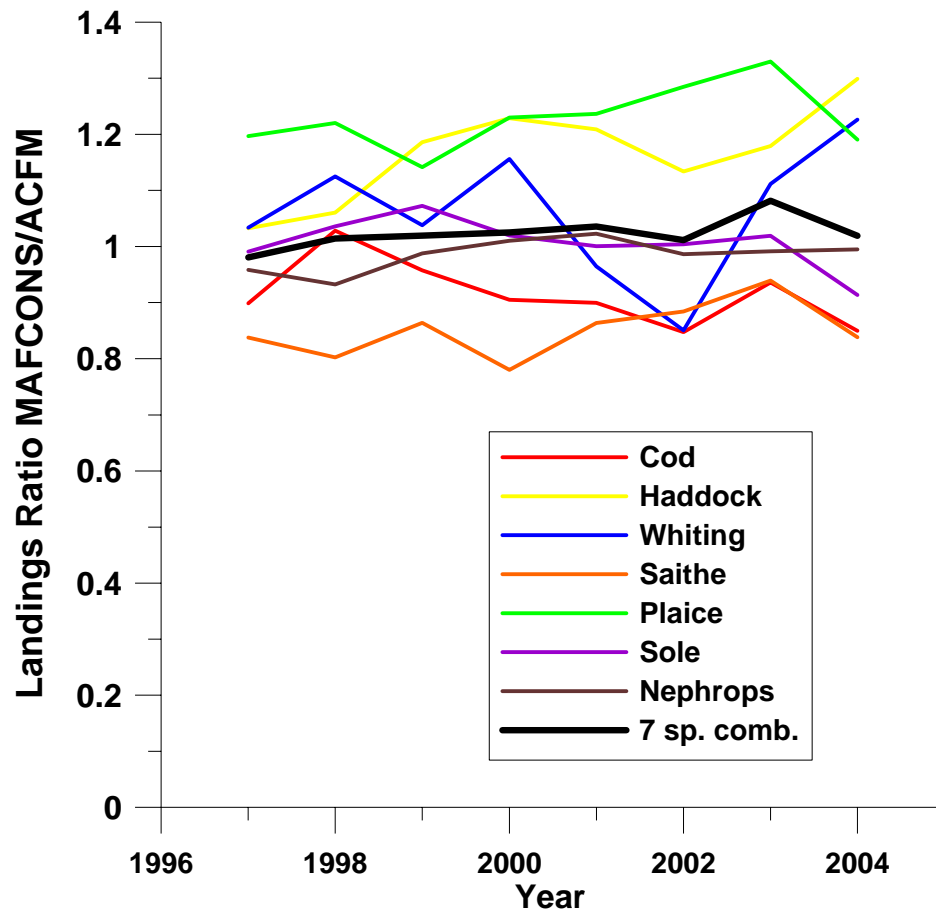


Figure 8.6.1.1.2. Annual variation in the ratio of total landings of each species estimated for the nine main North Sea fishing nations in the MAFCONS database and reported by ICES (2005) for these nations combined.

8.6.1.2. *Spatial distributions by gear category*

Cod was caught through out the North Sea in otter trawl directed at fish for human consumption, and in seine gear, particularly in the northeastern North Sea. Substantial amounts of cod were also taken in the beam trawl fisheries in the southeastern North Sea and in the otter trawl fishery directed at *Nephrops* (Figure 8.6.1.2.1). haddock were primarily taken in the two otter trawl fisheries and also in seine gear. Few were landed by beam trawlers (Figure 8.6.1.2.2). Whiting were primarily landed by otter trawlers targeting both fish for human consumption and *Nephrops*, and by seine fishing boats. However, substantial quantities were also landed by beam trawlers, particularly those operating in the extreme southern North Sea (Figure 8.6.1.2.3). Saithe were almost exclusively landed by otter trawlers targeting fish for human consumption fishing in the deeper water in the northern and northeastern North Sea. However, small quantities were also landed by seiners. Landings by otter trawlers targeting *Nephrops* and beam trawlers were low (Figure 8.6.1.2.4). Plaice were taken primarily by beam trawlers in the southern half of the North Sea. But substantial catches were also made by otter trawlers through out the North Sea, especially those operating in the southern North Sea. Plaice were landed in significant quantities by otter trawlers targeting *Nephrops* in the two central North Sea *Nephrops* grounds. Seiner also landed some plaice, particularly from rectangles close to the southern limits of seine activity (Figure 8.6.1.2.5). Sole were mainly landed by beam trawlers operating in the southern North Sea. Otter trawlers operating in the southern North

Sea also landed sole, but in much smaller quantities. Sole were rarely taken in seine gear. (Figure 8.6.1.2.6). *Nephrops* were primarily caught in otter trawl operations directed specifically at *Nephrops*, mainly in the western North Sea, but were also caught by otter trawlers targeting fish for human consumption. Quantities of *Nephrops* taken in seine gears and beam trawls were much lower. (Figure 8.6.1.2.7). Spatial patterns for each species and gear were similar in both time periods, particularly when the overall reductions in cod and whiting landings were taken into account (Figures 8.6.1.2.1 to 8.6.1.2.7).

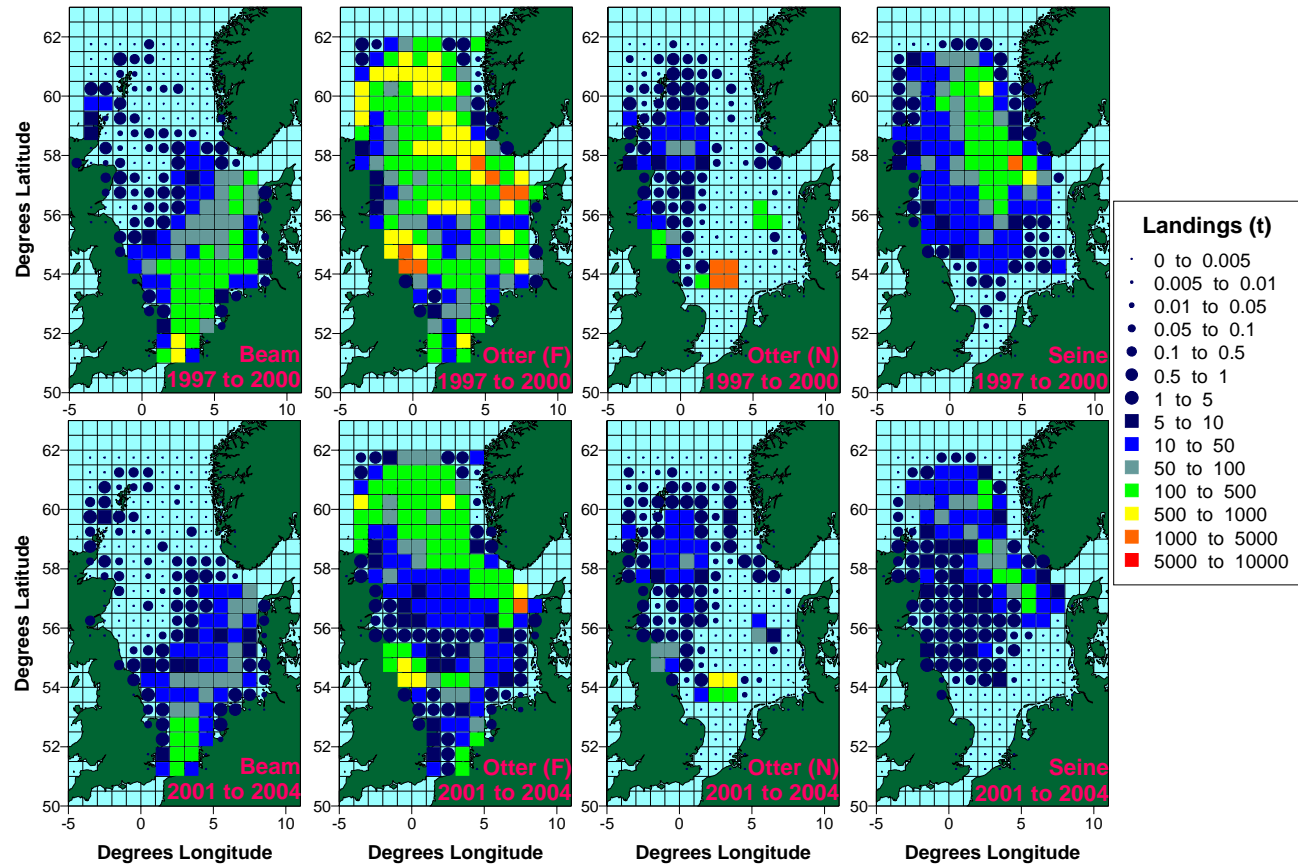


Figure 8.6.1.2.1. Spatial distributions in the average annual landings of cod taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by the nine major fishing nations operating in the North Sea in two four year periods, 1997 to 2000 and 2001 to 2004.

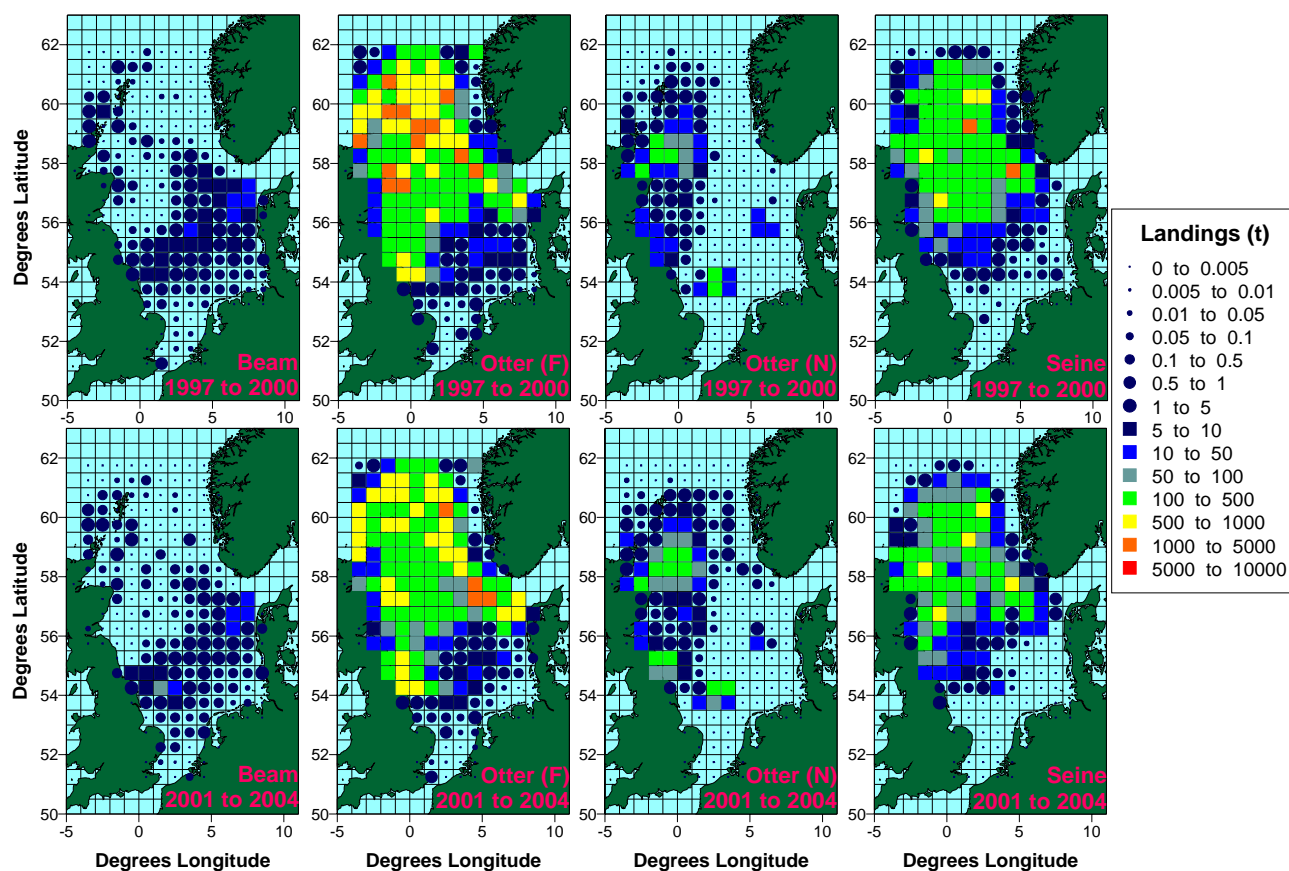


Figure 8.6.1.2.2. Spatial distributions in the average annual landings of haddock taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by the nine major fishing nations operating in the North Sea in two four year periods, 1997 to 2000 and 2001 to 2004.

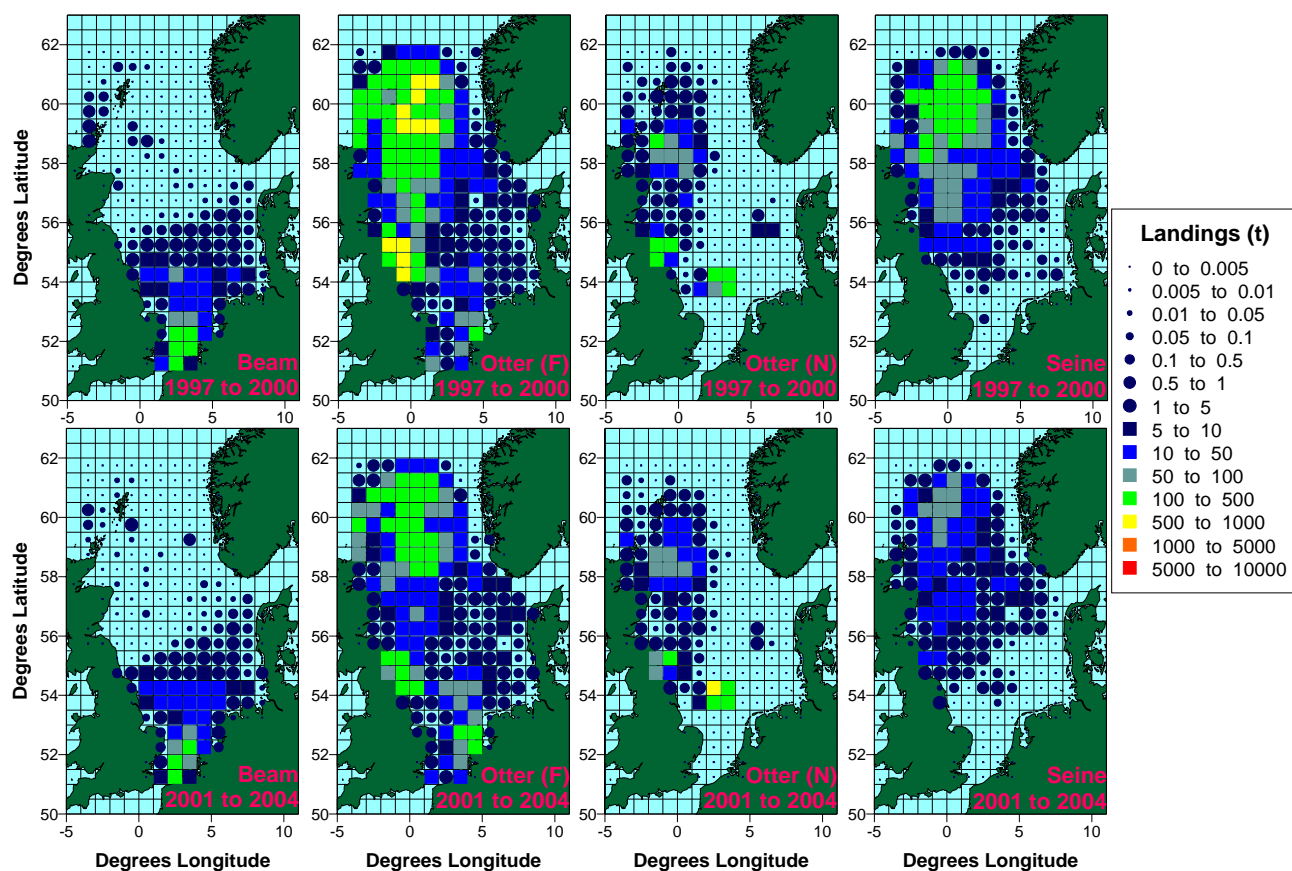


Figure 8.6.1.2.3. Spatial distributions in the average annual landings of whiting taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by the nine major fishing nations operating in the North Sea in two four year periods, 1997 to 2000 and 2001 to 2004.

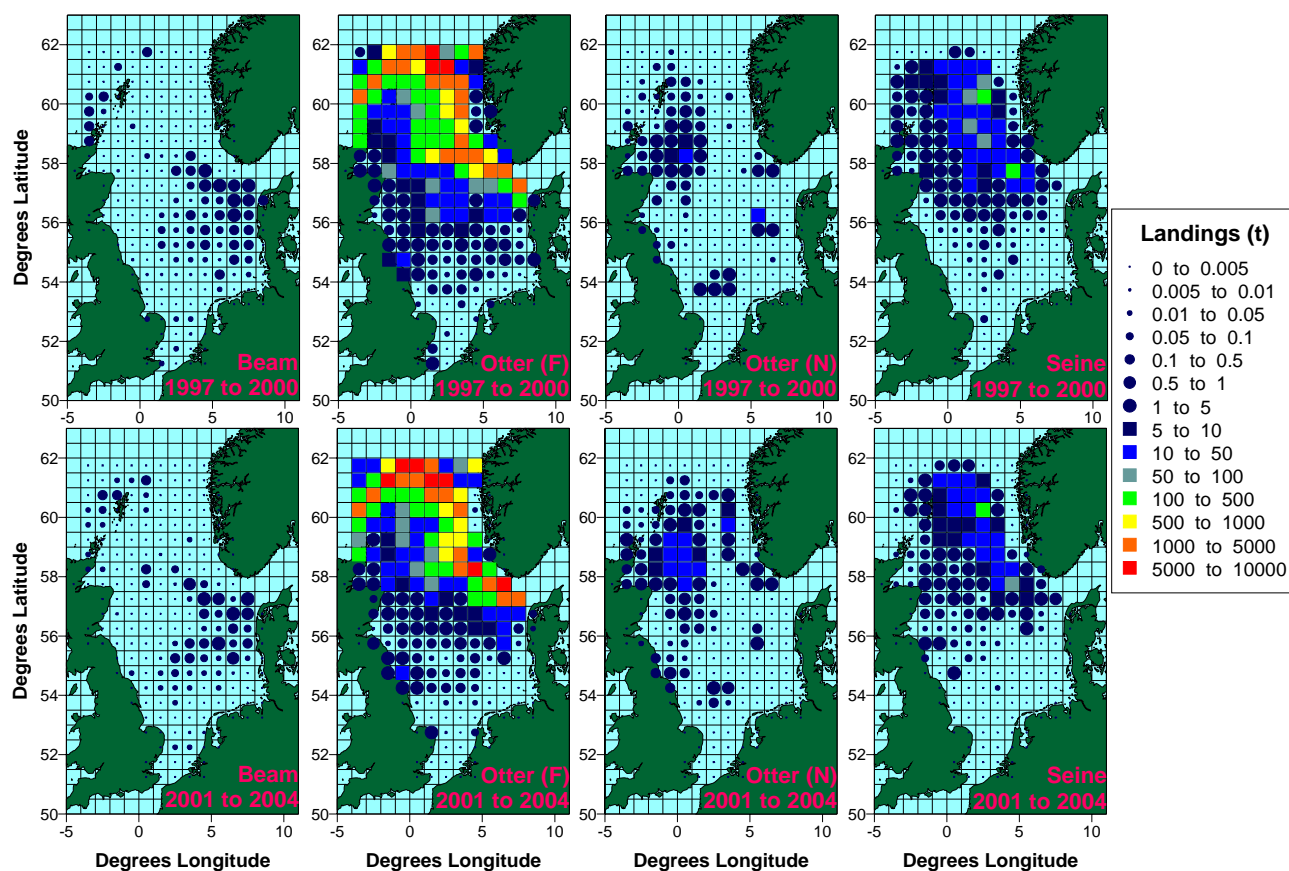


Figure 8.6.1.2.4. Spatial distributions in the average annual landings of saithe taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by the nine major fishing nations operating in the North Sea in two four year periods, 1997 to 2000 and 2001 to 2004.

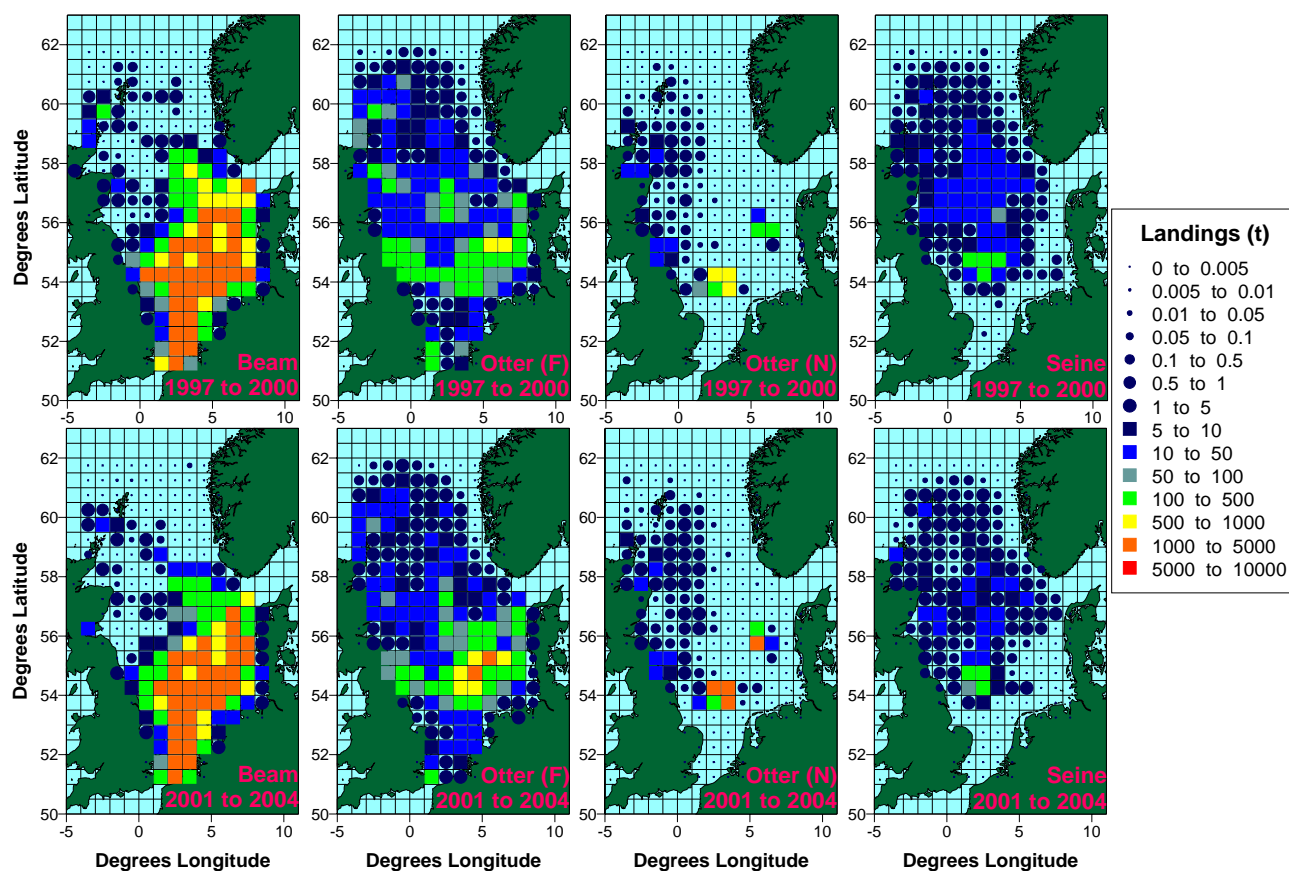


Figure 8.6.1.2.5. Spatial distributions in the average annual landings of plaice taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by the nine major fishing nations operating in the North Sea in two four year periods, 1997 to 2000 and 2001 to 2004.

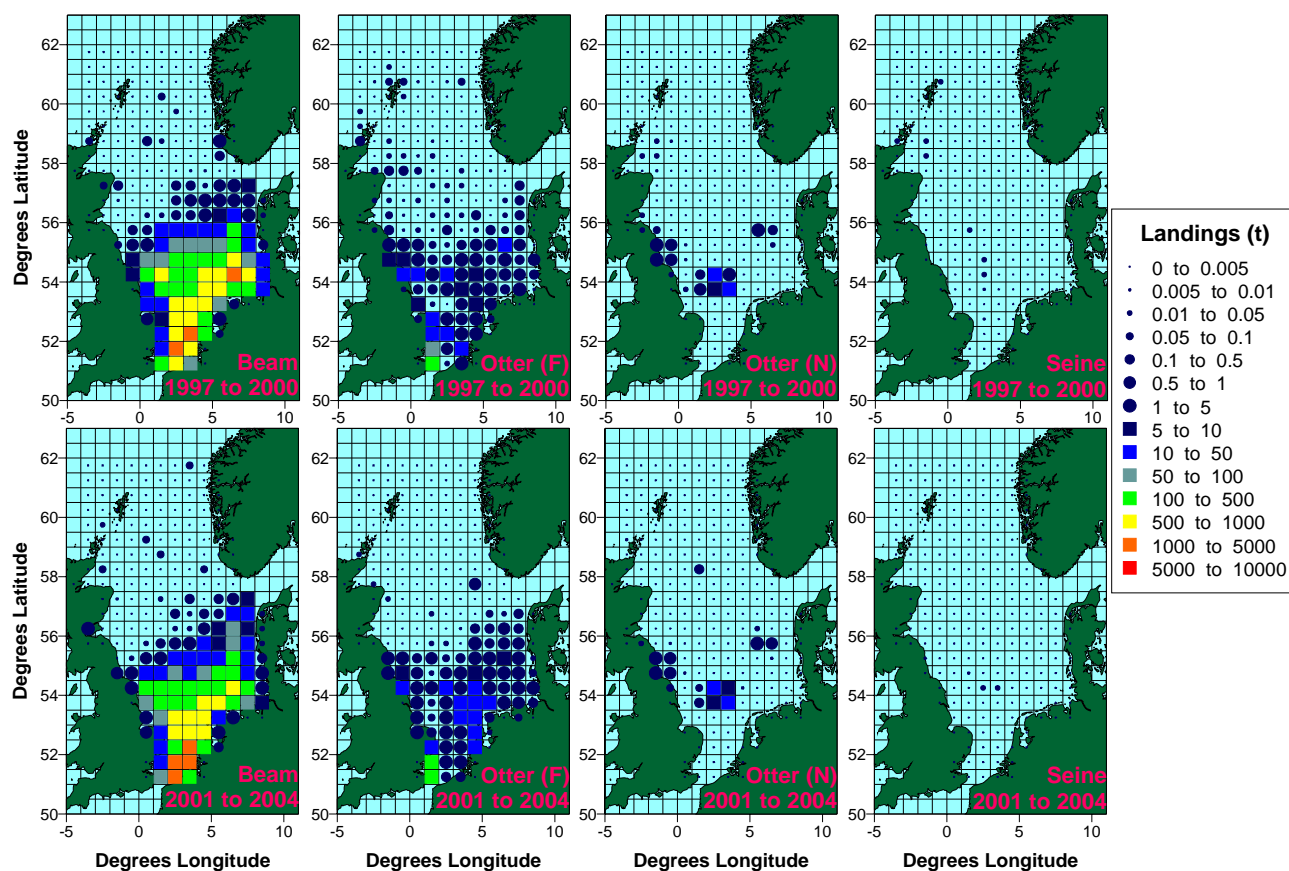


Figure 8.6.1.2.6. Spatial distributions in the average annual landings of sole taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by the nine major fishing nations operating in the North Sea in two four year periods, 1997 to 2000 and 2001 to 2004.

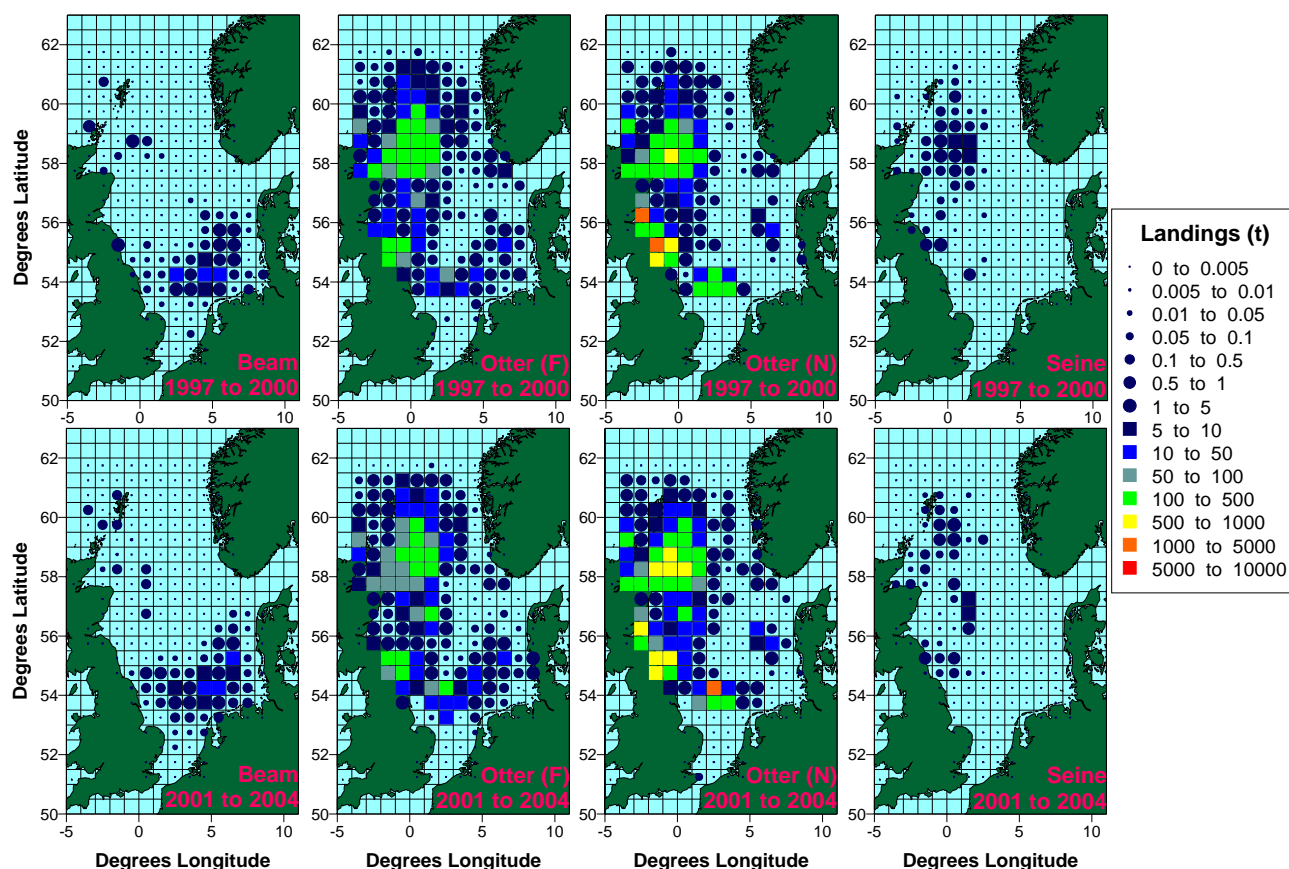


Figure 8.6.1.2.7. Spatial distributions in the average annual landings of *Nephrops* taken in four main gear categories, beam trawl, otter trawl directed at fish for human consumption {Otter(F)}, otter trawl directed at *Nephrops* {Otter(N)}, and seine gear by the nine major fishing nations operating in the North Sea in two four year periods, 1997 to 2000 and 2001 to 2004.

8.6.1.3. *Species spatial distributions (all gears combined)*

Spatial distributions of total landings of each species in each year are presented in Figures 8.6.1.3.1 to 8.6.1.3.7. Cod landings show two distinct “hotspots”, in the northeastern and in the southwestern North Sea. Over time, the trough between these concentrations deepened as cod landings in general declined (Figures 8.6.1.3.1). In the late 1990s, haddock were landed from throughout the northern half of the North Sea, but as time progressed two main areas emerged, one off northeast Scotland and one along the edge of the Norwegian deeps (Figures 8.6.1.3.2). Whiting landings predominantly originated from two main regions, one in the northern North Sea and the second in the southwestern North Sea. Over time, productivity per rectangle in the northern “hotspot” declined more than it did in the southern region (Figures 8.6.1.3.3). Saithe were primarily landed from the extreme northern and northeastern edges of the North Sea in all years (Figures 8.6.1.3.4). Plaice landings were mainly taken throughout most of the southern half of the North Sea, with little change in distribution over time (Figures 8.6.1.3.5). In all years sole landings were highest from the very extreme southern North Sea (Figures 8.6.1.3.6). *Nephrops* landings distributions also changed little over time coming predominantly from the main *Nephrops* grounds in the Fladden area, along the east coast of the UK, and from the Botney Gut and Horns Reef regions in the central and east central North Sea (Figures 8.6.1.3.7).

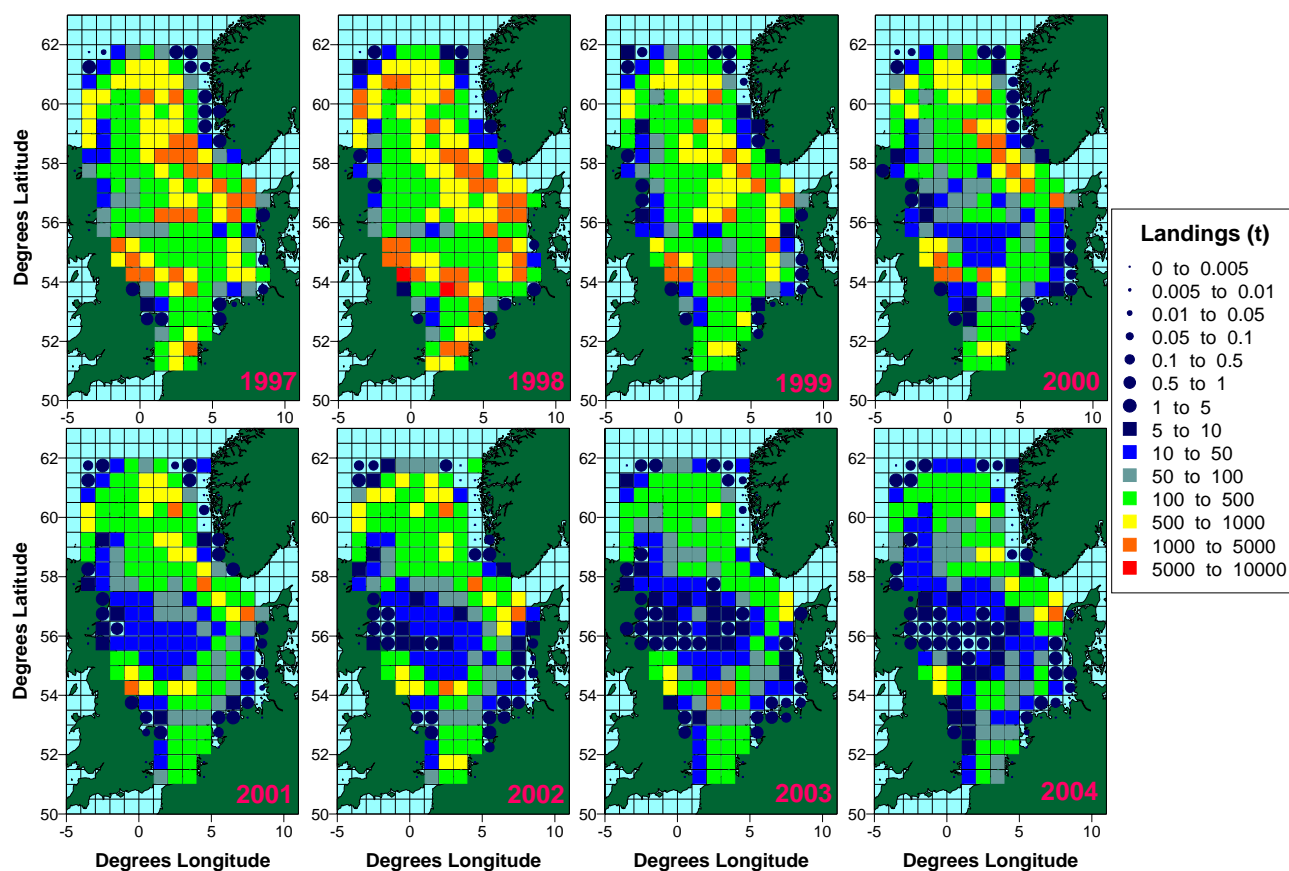


Figure 8.6.1.3.1. Spatial distributions of cod landings across all fishing gears by the nine major fishing nations operating in the North Sea in each of the years between 1997 and 2004.

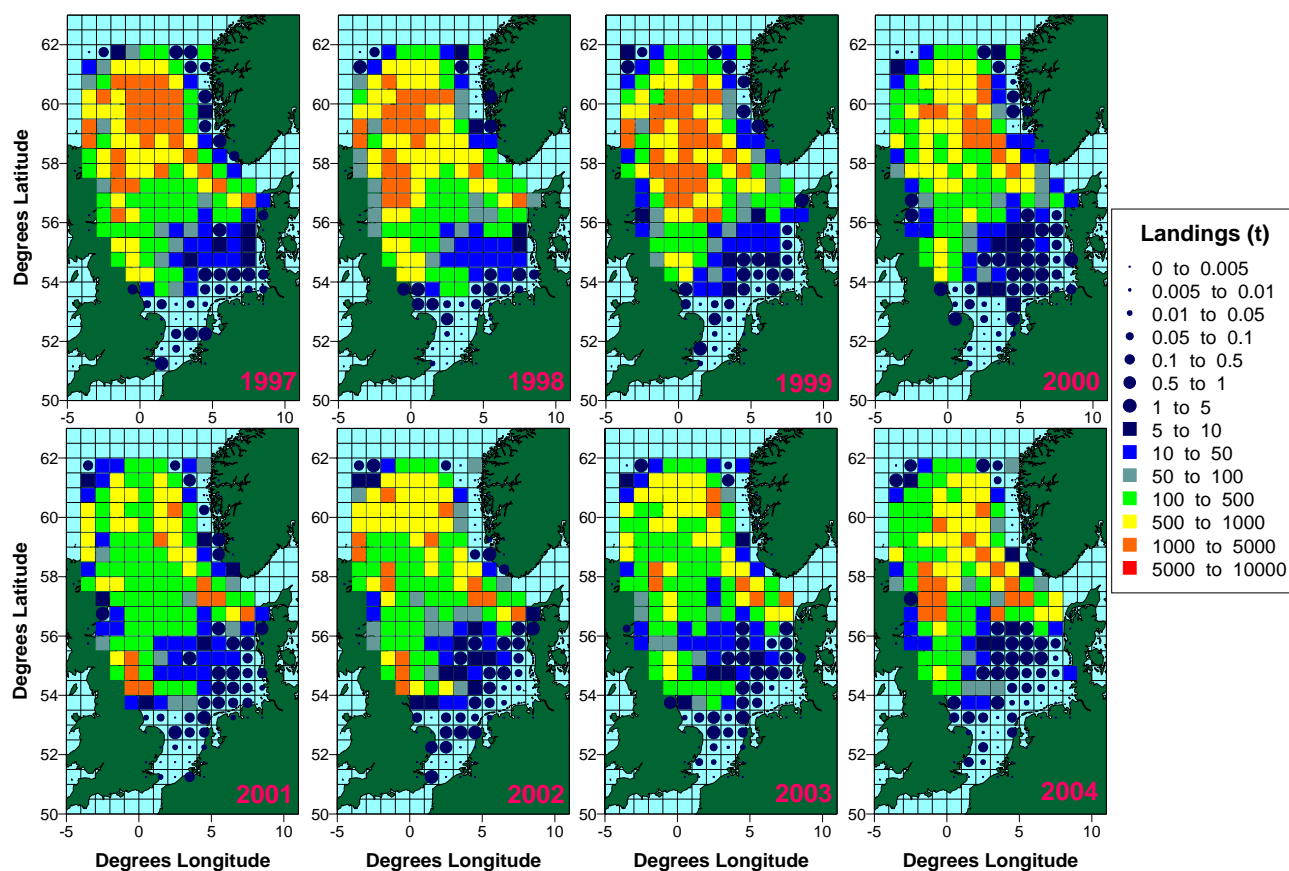


Figure 8.6.1.3.2. Spatial distributions of haddock landings across all fishing gears by the nine major fishing nations operating in the North Sea in each of the years between 1997 and 2004.

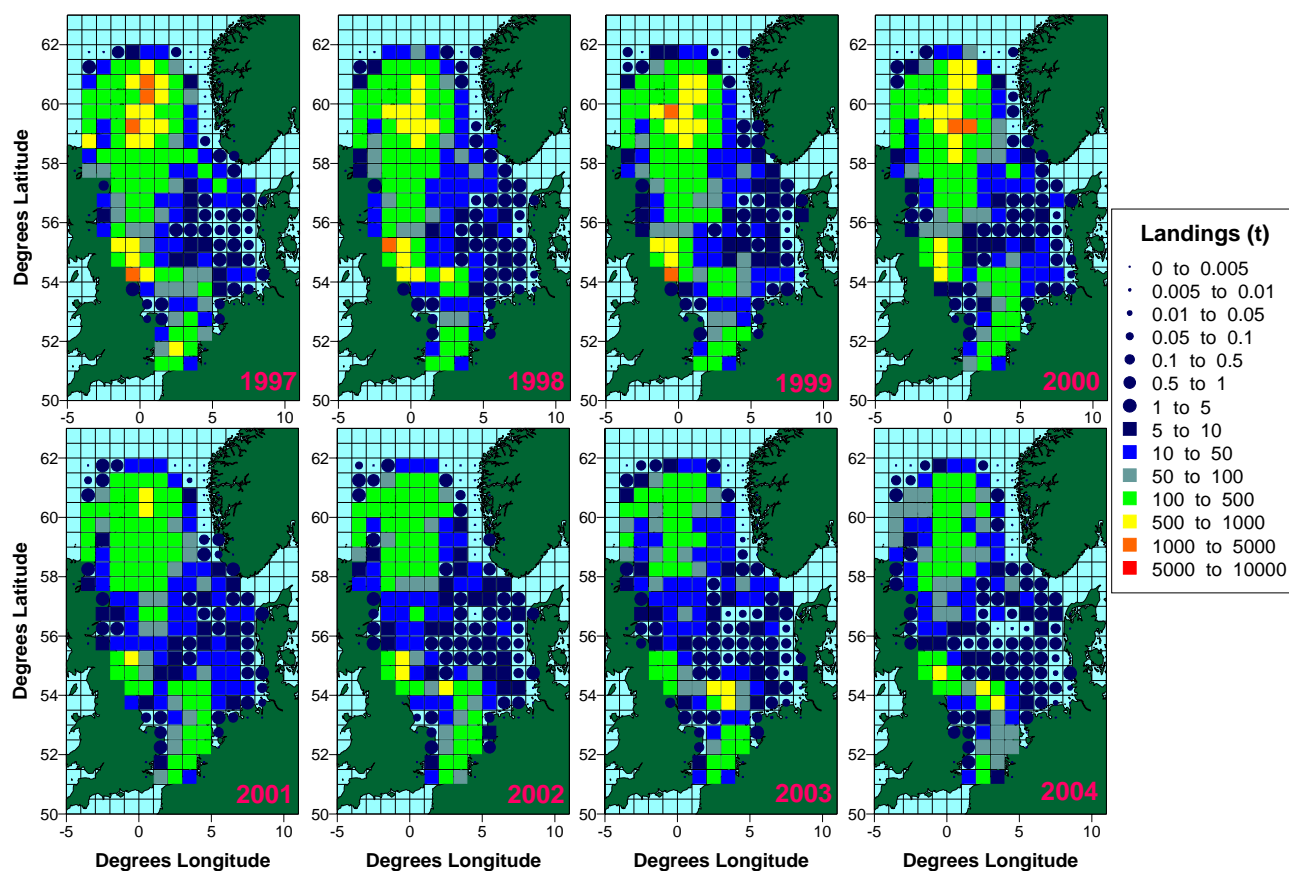


Figure 8.6.1.3.3. Spatial distributions of whiting landings across all fishing gears by the nine major fishing nations operating in the North Sea in each of the years between 1997 and 2004.

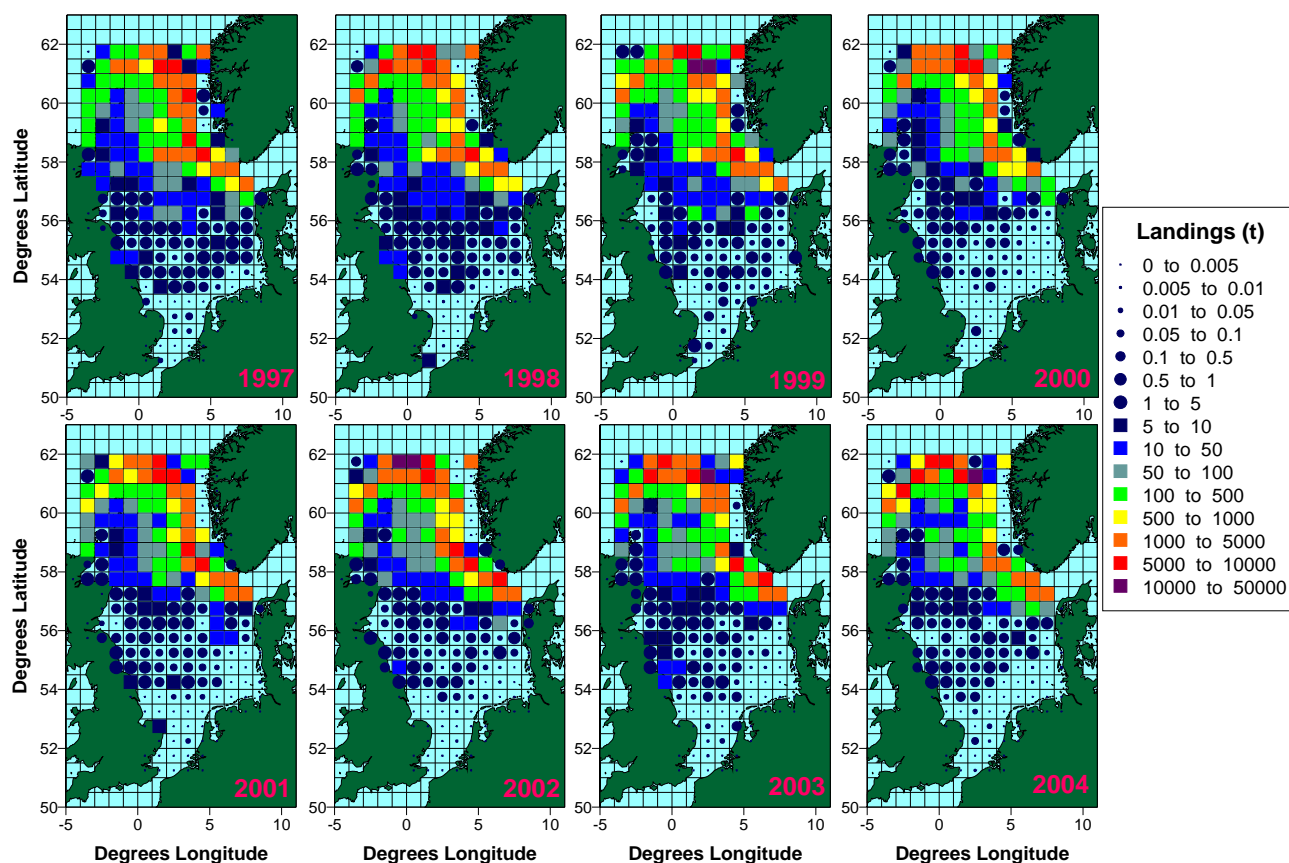


Figure 8.6.1.3.4. Spatial distributions of saithe landings across all fishing gears by the nine major fishing nations operating in the North Sea in each of the years between 1997 and 2004.

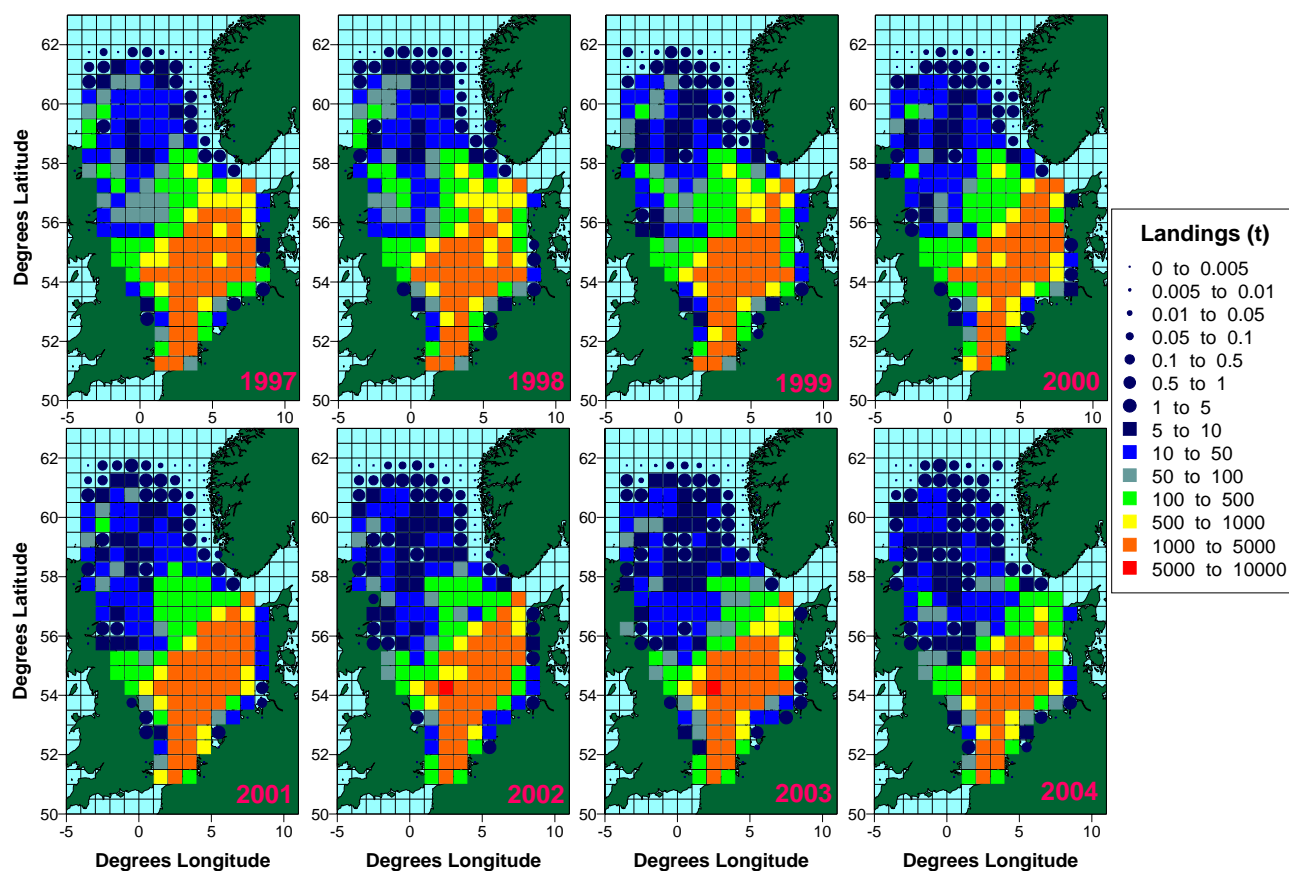


Figure 8.6.1.3.5. Spatial distributions of plaice landings across all fishing gears by the nine major fishing nations operating in the North Sea in each of the years between 1997 and 2004.

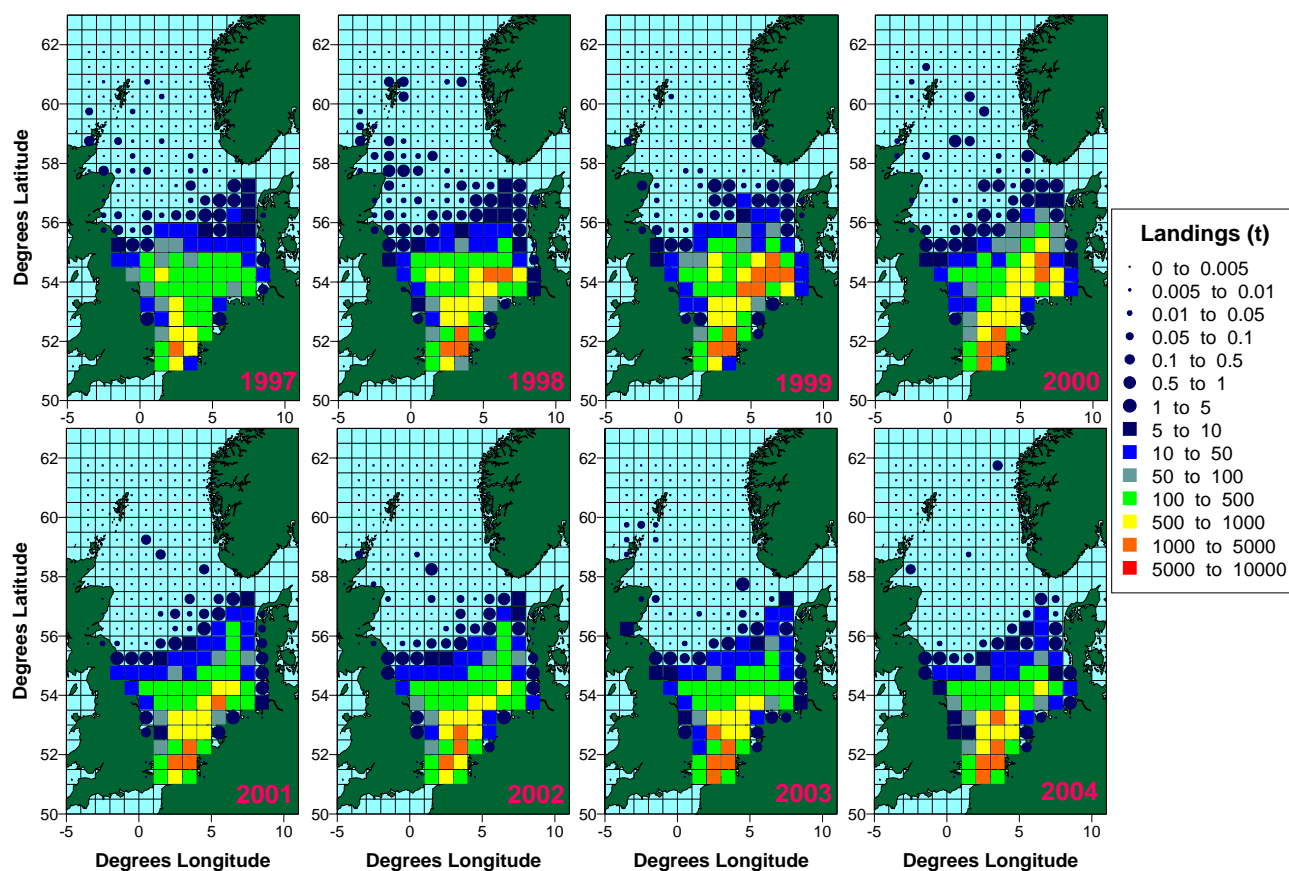


Figure 8.6.1.3.6. Spatial distributions of sole landings across all fishing gears by the nine major fishing nations operating in the North Sea in each of the years between 1997 and 2004.

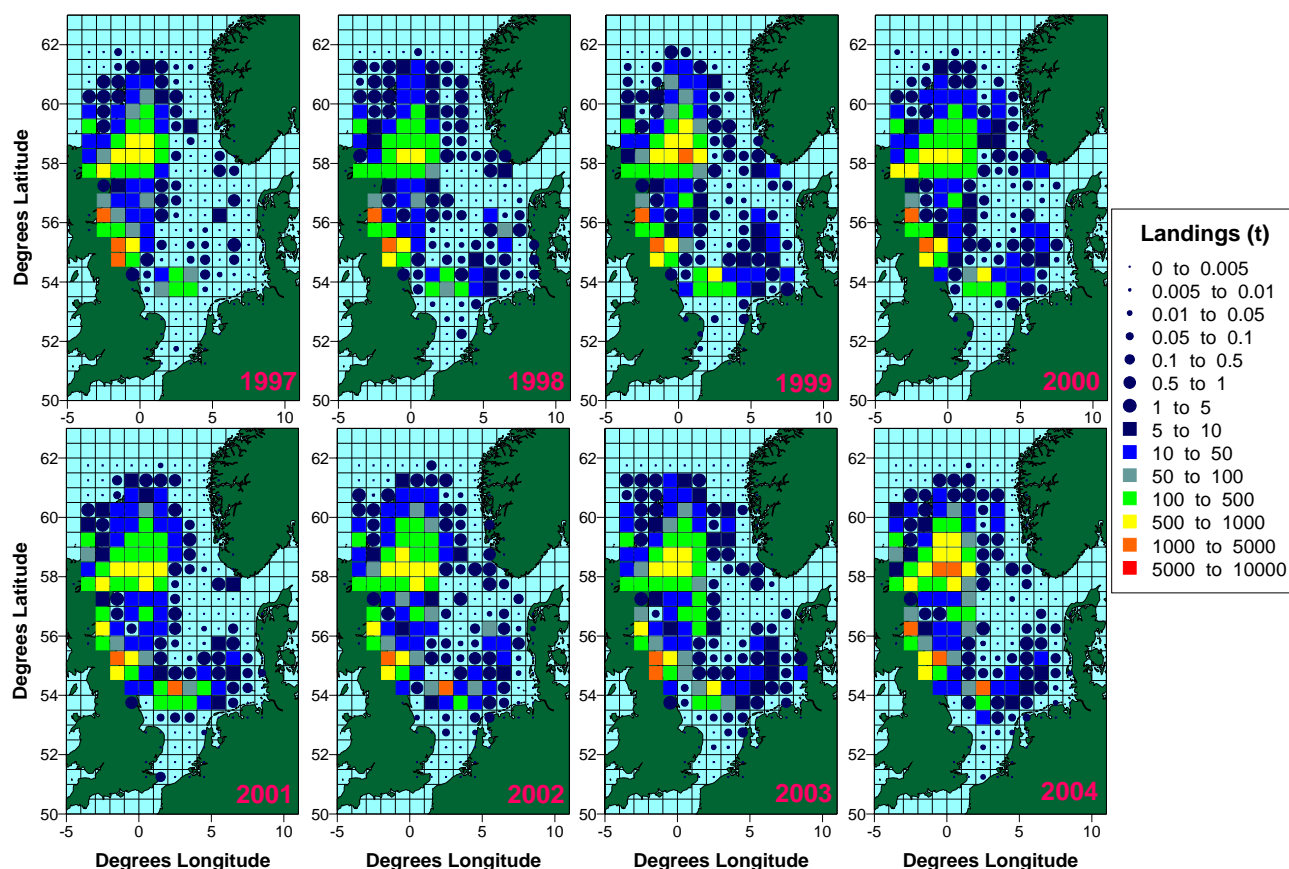


Figure 8.6.1.3.7. Spatial distributions of *Nephrops* landings across all fishing gears by the nine major fishing nations operating in the North Sea in each of the years between 1997 and 2004.

8.6.2. Effort

8.6.2.1. *Annual totals*

Over the period 1997 to 2004, the combination of both the effort data supplied by institutes participating in the MAFCONS project and the modelled estimates of effort for the four non-MAFCONS countries suggest that overall fishing effort declined by 28% (Figure 8.6.2.1.1). However this reduction in fishing effort was not evenly distributed across all fishing gears and fishing nations. While beam trawl effort, otter trawl effort directed at fish and seine gear effort all declined by 31%, 44%, and 62% respectively, otter trawl effort directed at *Nephrops* increased by 65%. More or less linear declines in total fishing effort were apparent for the UK (England, Wales and Northern Ireland) (55%), UK (Scotland) (45%), Belgium (38%), Sweden (35%), Netherlands (22%), and Germany (21%).

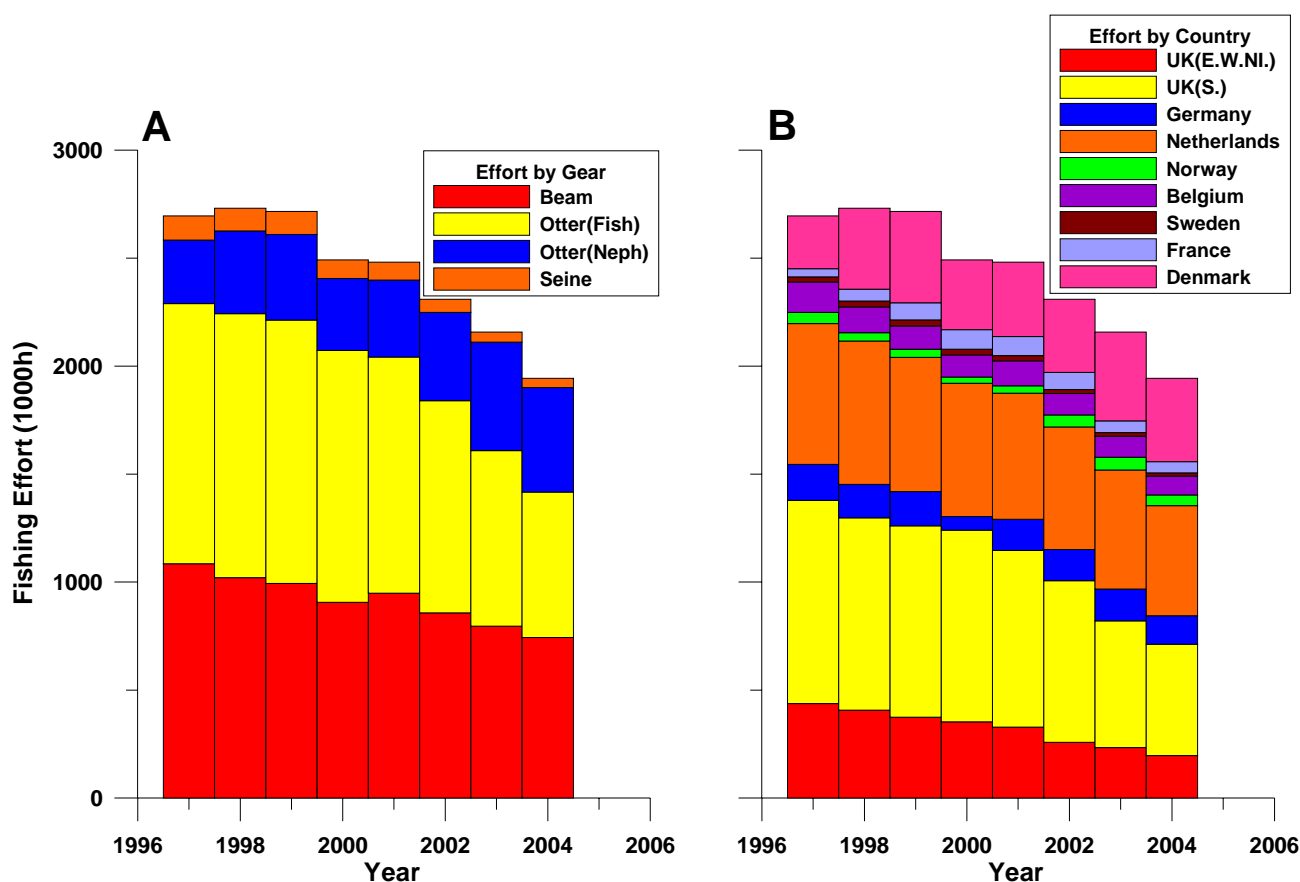


Figure 8.6.2.1.1. Trends in total annual fishing effort by the nine nations with major fisheries in the North Sea, combining reported data for the five MAFCONS participating countries (UK: England, Wales, and Northern Ireland {UK(E.W.NI.)}, UK: Scotland {UK(S.)}, Germany, The Netherlands and Denmark) and the model estimated data for the four non-participating countries (Belgium, Sweden, France and Denmark), over the period 1997 to 2004. A. disaggregated by main gear category. B. disaggregated by the nine countries.

8.6.2.2. Spatial distributions by gear category

When combining the effort data supplied by the five MAFCONS countries with the data modelled for the four non-MAFCONS countries, the basis spatial patterns observed for each gear for the data reported by the MAFCONS countries alone still emerged. Beam trawling was essentially a southern North Sea activity with small amounts occurring in the extreme northwestern North Sea. No real change in the spatial distribution of beam trawl effort occurred over the time period (Figure 8.6.2.2.1). Otter trawling directed at fish for human consumption occurred through out the North Sea, but effort in the north was considerably higher than in the south. Over time however, otter trawl activity directed at fish for human consumption declined more in this northern sector than in the south (Figure 8.6.2.2.2). Not surprisingly, otter trawl effort directed at *Nephrops* was concentrated around the main *Nephrops* grounds in the North Sea in all eight years (Figure 8.6.2.2.3). Fishing with seine gears occurred mainly in the northern North Sea, and little change in the distribution was apparent over time (Figure 8.6.2.2.4).

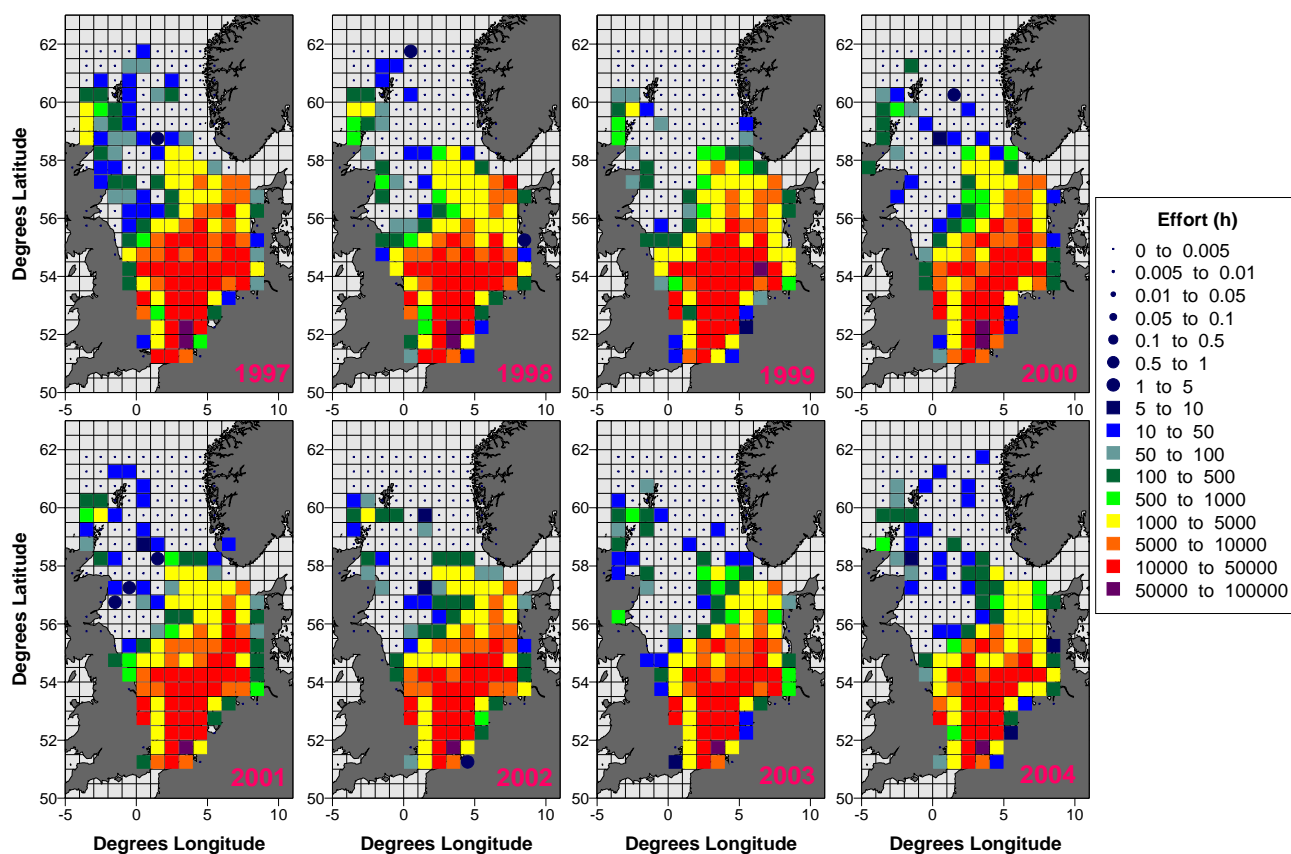


Figure 8.6.2.2.1. Spatial distributions of fishing effort (hours-fishing) using beam trawl by the nine major fishing nations operating in the North Sea in each of the years between 1997 and 2004.

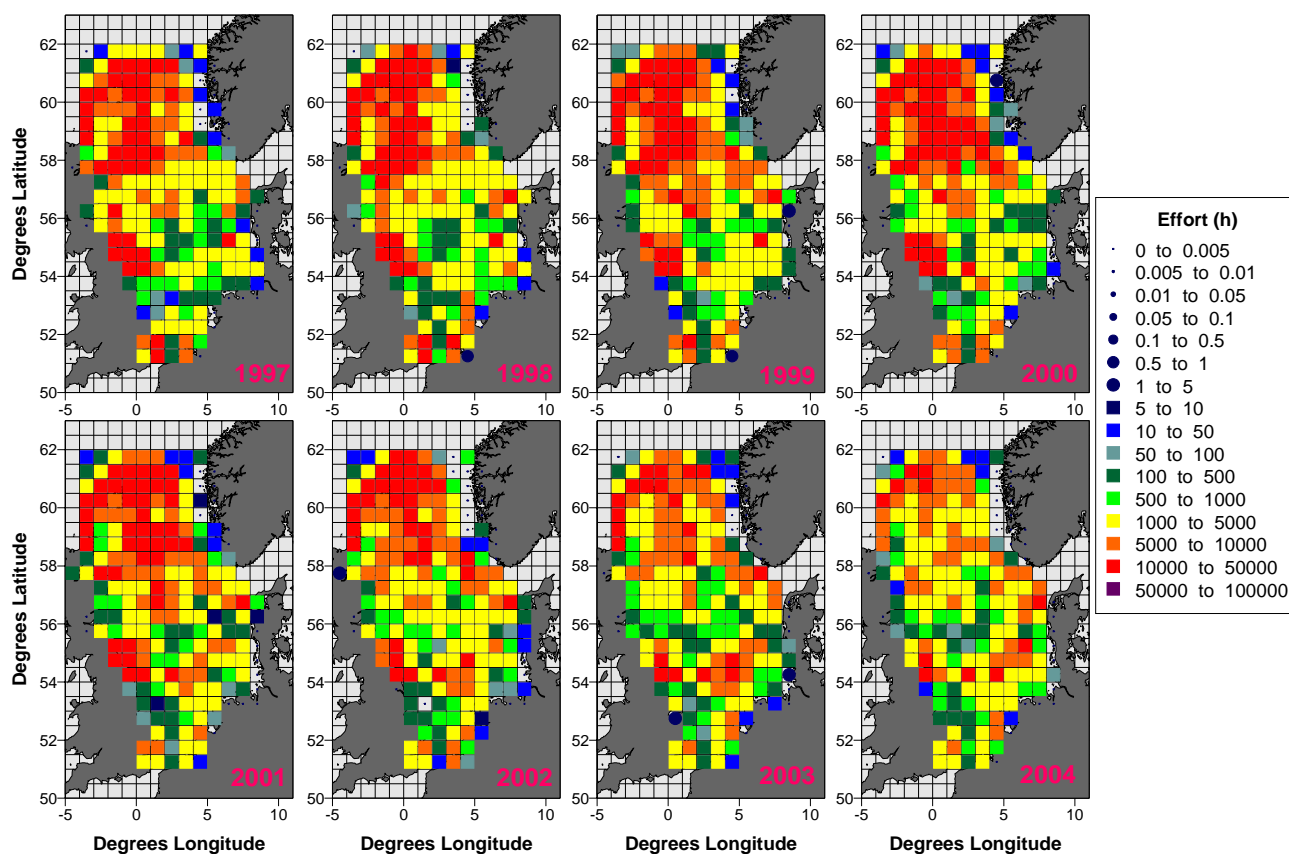


Figure 8.6.2.2.2. Spatial distributions of fishing effort (hours-fishing) using otter trawl directed at fish for human consumption by the nine major fishing nations operating in the North Sea in each of the years between 1997 and 2004.

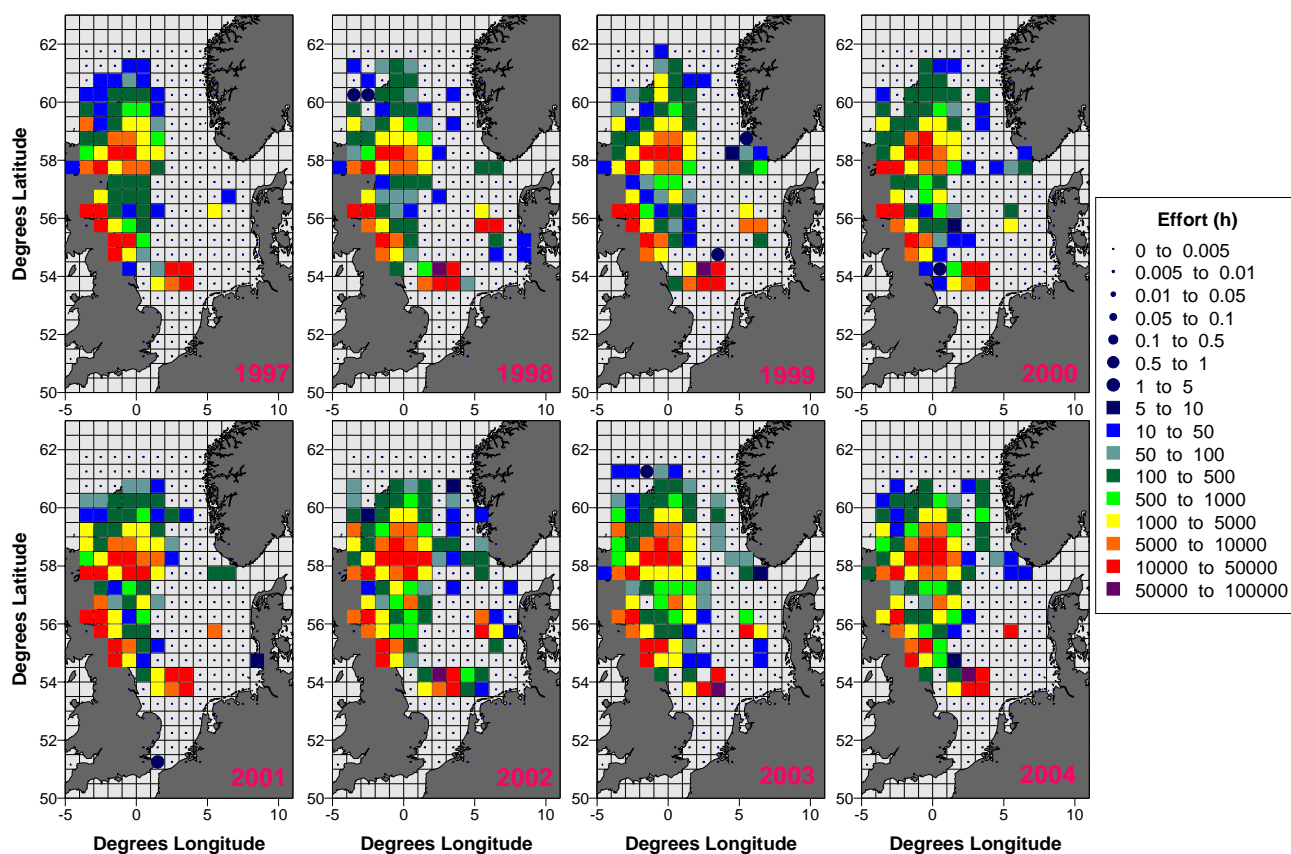


Figure 8.6.2.2.3. Spatial distributions of fishing effort (hours-fishing) using otter trawl directed at *Nephrops* by the nine major fishing nations operating in the North Sea in each of the years between 1997 and 2004.

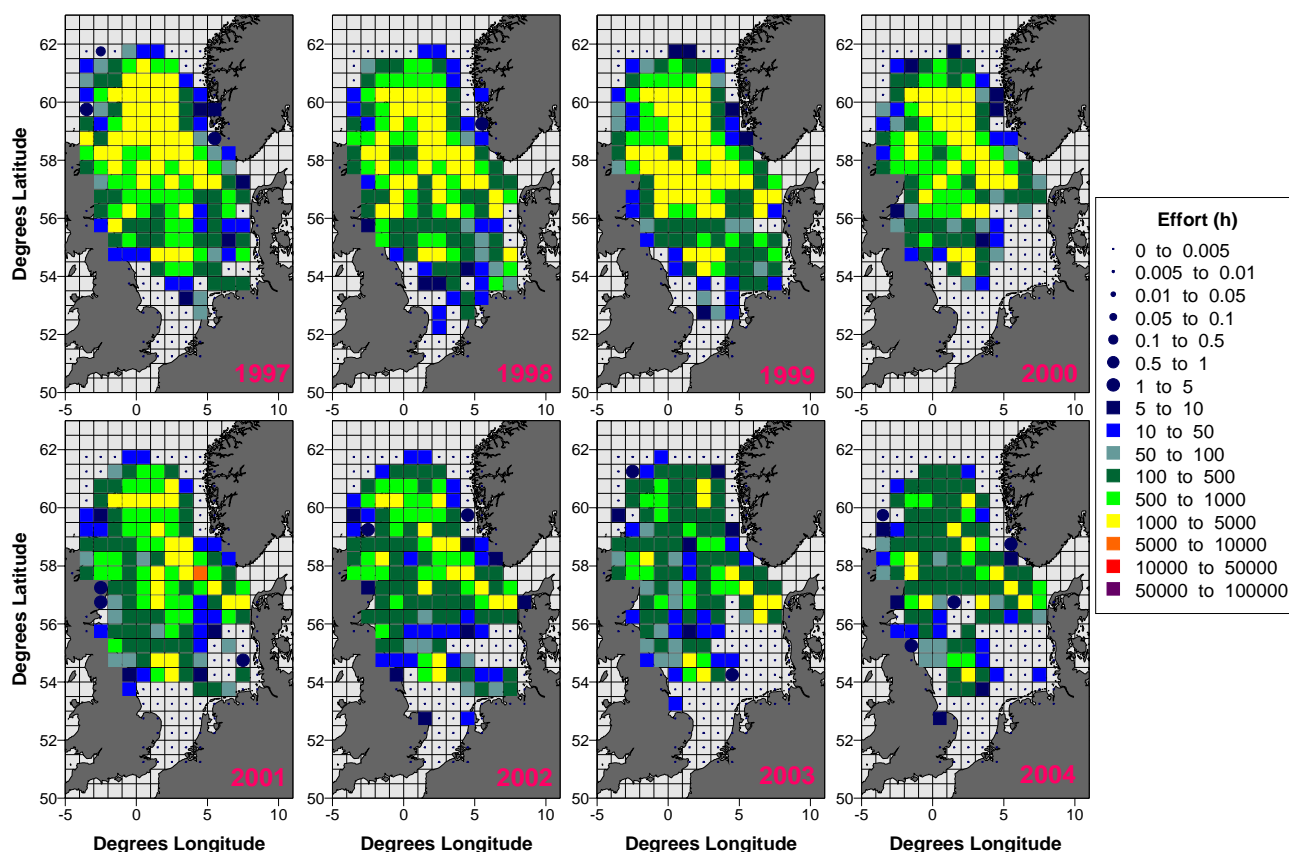


Figure 8.6.2.2.4. Spatial distributions of fishing effort (hours-fishing) using seine gear by the nine major fishing nations operating in the North Sea in each of the years between 1997 and 2004.

8.7. Modelling the mortality of benthic invertebrates

8.7.1. Underlying rationale

If a constant proportion of animals is killed by each passage of a fishing gear, such that this proportion is killed on the first fishing event, the second fishing then kills the same proportion of the animals that survived the first fishing, the third fishing then 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 section 8.4. 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 would be 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, $0.8^5\%$ (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 (approximatel 1NM by 1 Nm) 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 et al 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 in press). 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.

8.7.2. Model development

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 et al 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, with notation adapted for our particular circumstances, is described by the following equation (Pollard 1977):

$$P(N_{SU}) = \frac{e^{-\bar{x}} \bar{x}^{N_{SU}}}{N_{SU}!} \quad 8.7.2.1.$$

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 sub-units in the ICES rectangle is \bar{x} . To calculate these

probabilities for each of the sub-units in any specific ICES rectangle, it is first necessary to estimate the mean number of tows across all sub-units in the rectangle. This is simply done by:

$$\bar{x} = \frac{T_{Rect} / T_{Tow}}{RECT_{SU}} \quad 8.7.2.2.$$

where T_{Rect} is the total number of hours fishing recorded in the ICES rectangle, T_{Tow} is the average tow duration and $RECT_{SU}$ is the number of sub-units in the ICES rectangle. Substituting equation 8.7.2.2 into equation 8.7.2.1, 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:

$$P(N_{SU}) = \frac{e^{-\left(\frac{T_{Rect}/T_{Tow}}{RECT_{SU}}\right)} \left(\frac{T_{Rect}/T_{Tow}}{RECT_{SU}}\right)^{N_{SU}}}{N_{SU}!} \quad 8.7.2.3.$$

The number of sub-units with all possible numbers of tows can be calculated by multiplying these individual probabilities by the number of sub-units in each ICES rectangle ($RECT_{SU}$).

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:

$$FF_{SU} = A_F / A_{SU} \quad 8.7.2.4.$$

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:

$$A_F = N_{SU} * T_{Tow} * V_{Tow} * W_G \quad 8.7.2.5.$$

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 [$\times 1.853 = 55.59\text{km}$]) 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:

$$A_{SU} = \frac{30 * 60 * 1.853^2 * \cos(Lat_{rect})}{RECT_{SU}} = 6.867218 * \cos(Lat_{rect}) \quad 8.7.2.6.$$

Substituting equations 8.7.2.5 and 8.7.2.6 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:

$$FF_{SU} = \frac{N_{SU} * T_{Tow} * V_{Tow} * W_G}{6.867218 * \cos(Lat_{rect})} \quad 8.7.2.7.$$

Knowing the frequency that each ICES rectangle sub-unit has been fished on average, and with information regarding mortality per fishing tow (eg. Collie et al 2000; Kaiser et al 2006), 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:

$$M_{Total} = 1 - e^{-Ln[1-M_{Tow}] * FF_{SU}} \quad 8.7.2.8.$$

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 individual fishing tows distributed across the rectangle following a Poisson distribution, this is calculated by substituting equations 8.7.2.3 and 8.7.2.7 into equation 8.7.2.8, summing over all sub-units and dividing by the number of sub-units in the rectangle, $RECT_{SU}$:

$$M_{Rect} = \frac{\sum_{SU=1}^{900} 1 - e^{-Ln[1-M_{Tow}] * \left(\frac{e^{-\left(\frac{T_{Rect}/T_{Tow}}{RECT_{SU}}\right) * \left(\frac{T_{Rect}/T_{Tow}}{RECT_{SU}}\right)^{N_{SU}}}}{N_{SU}!} * T_{Tow} * V_{Tow} * W_G \right)}}{RECT_{SU}} \quad 8.7.2.9.$$

For the purpose of this study, and following precedent set by Rijnsdorp et al (1998) and Piet et al (2000), we sub-divide each ICES rectangle into 900 sub-units. For ICES rectangles with 100% sea area, $RECT_{SU}$ therefore equals 900. However, for coastal ICES rectangles the proportion of the rectangle consisting of sea-cover was used to adjust the number of sub-units. Thus, for example, if an ICES rectangle had only 50% sea-cover, the number of sub-units ($RECT_{SU}$) was reduced to 450. Consequently sub-unit area within rectangles was kept independent of sea cover. Because of this, in equation 8.7.2.6, $RECT_{SU}$ was always maintained at 900. The non-linear relationship between the measure of fishing activity ($\text{hrs.yr}^{-1}.\text{rect}^{-1}$) derived from this model is demonstrated in Figure 8.7.2.1 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 900 sub-units in the full rectangle, assumes a rectangle with a mid-point latitude of 54.75°N.

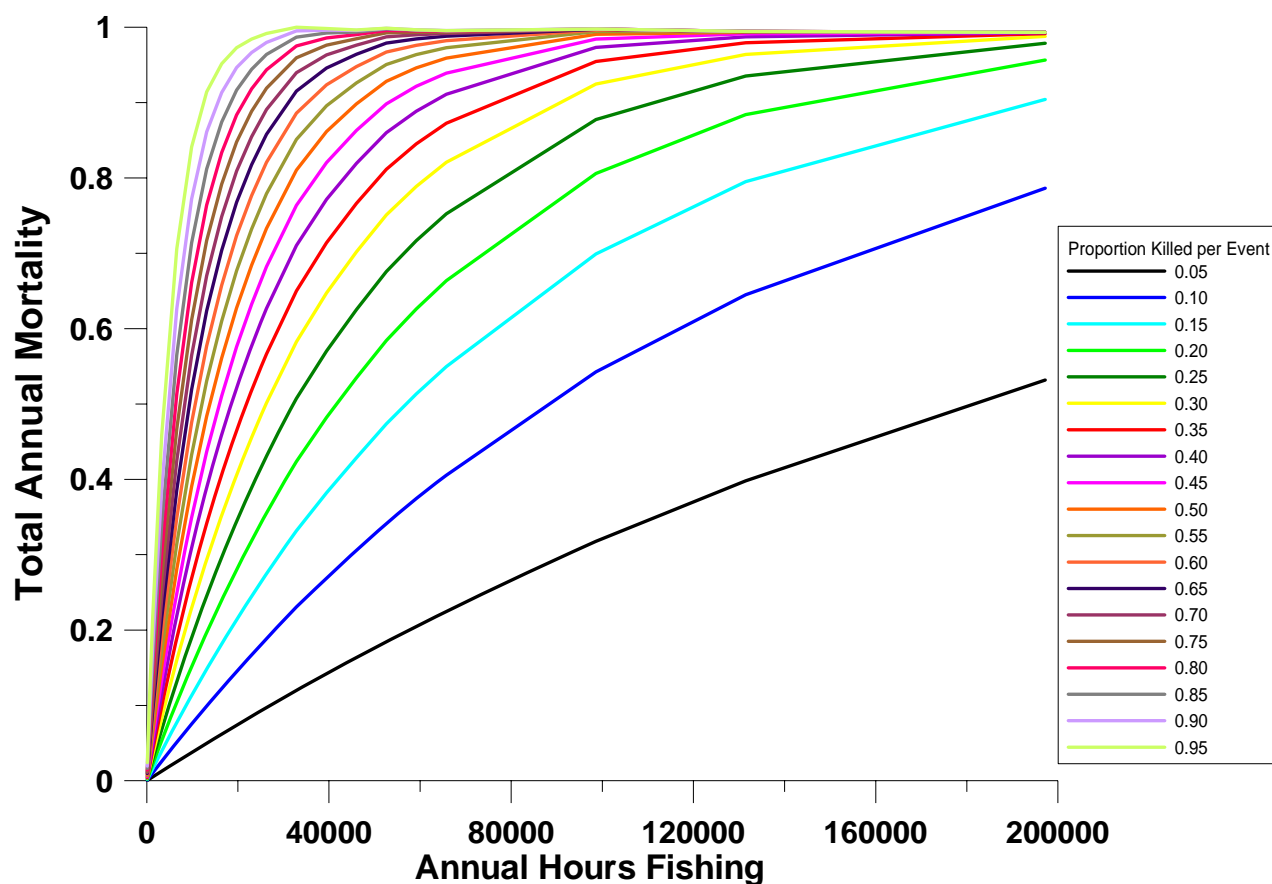


Figure 8.7.2.1. Relationship between total beam trawl fishing effort in an ICES rectangle and the resultant total mortality of resident benthic invertebrates at various “per fishing event” mortality rates.

8.7.3. The distribution of fishing activity

The most critical input parameter into the model is the fishing activity information. The model requires data at ICES statistical rectangle scale. Two EC projects have now attempted to assemble “international” effort databases in order 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 current “MAFCONS” project assimilated data for the period 1997 to 2004. The MAFCONS project aggregated data for 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 this database therefore only includes data from the other five countries mentioned. 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. The MAFCONS project however “modelled” fishing effort distributions for the four main “missing” countries that have significant fishing interests in the North Sea (see Section 8.5.2), and the effort

data we use here to model benthic mortality are estimates derived from the data supplied by the five MAFCONS countries combined with the modelled estimates for the four non-MAFCONS countries.

8.7.4. Estimating mortality rates per fishing event

Per fishing event mortality rate data were estimated following Tulp et al (2006), based on information provided by the meta-analyses carried out by Collie et al. (2000) and Kaiser et al (2006). A selection was made from the meta-analysis of Kaiser *et al.* (2006) to extract Phyla level mortality rates (defined here as % change in population abundance) for all species in the epifauna invertebrate database. It was not possible to work at a higher taxonomic resolution because of the high variance and low numbers of studies that was increasingly found as taxonomic resolution increased. The meta-analysis database, constructed during the EC 5th framework project COST-IMPACT, was kindly supplied by Mike Kaiser and Hilmar Hinze of the University of Wales – Bangor. It includes the results of 101 different experimental manipulations or observations of the effects of fishing disturbance on benthic fauna and communities, extracted from 55 separate publications.

The selection used here was limited to studies that were carried out in temperate latitudes in the sub-tidal zone with otter trawl and beam trawl gears. Data were not available for seine gears and did not differentiate between otter trawls targeted at fish versus otter trawls targeted at invertebrates. It is assumed here that the mortalities associated with seine gears are half those found for otter trawls for fish and that the mortalities associated with otter trawls targeted at invertebrates are 1.25 times those associated with otter trawls targeted at fish because of the gear modifications that more actively disturb the seafloor and because the codend mesh size is reduced specifically to retain the smaller bodied *Nephrops*, which is itself a benthic invertebrate. Where beam trawl mortality was not available this was taken as 1.33 times the mortality of otter trawl targeted at fish and where otter trawl targeted at fish was not available this was taken as 0.75 times the beam trawl mortality. These are average ratios for the taxa for which data were available. For a number of Phyla there were no data. These were assigned an average gear mortality based on all other Phyla.

The selection limited the studies to those that only had one discrete disturbance event to reflect actual encounter mortality and a further selection was made based on the reported time in days sampled after the disturbance incidence (<2 days). We could not take different levels of background disturbance into account because this would reduce the dataset too severely. Also, we consider it important that the variability associated with differing levels of background disturbance should be included, to reflect the real situation in the North Sea. It is well known that mortality rates are also highly dependent on habitat characteristics (Kaiser *et al.* (2006)). Strictly speaking the mortality rates should be extracted for each different habitat type separately. However, many of the benthos species are very highly specific in their habitat preferences. In addition, the resolution at which habitat sediment type data are available is probably at too broad a resolution to accurately reflect the distributions of sediments within each ICES rectangle.

The magnitude of the response variable (% change in population abundance) was calculated from the following equation, using the mean values for fished and un-fished plots in any given study:

$$\% \text{difference} = ((A_f - A_c) / A_c) \times 100 \quad 8.7.4.1.$$

Where A_f is the abundance in fished plots and A_c is the abundance in un-fished control plots. For cases in which the study involved a before fishing-after fishing comparison for the same plot(s),

rather than a Y=Treatment-Control design, these data were used to calculate % difference by comparison of the pre-fishing treatment (A_c) with the post-fishing treatment (A_f)

8.7.5. Modelling the mortality caused by fishing

8.7.5.1. *Beam trawling*

The model was run using the annual average hours-fishing per ICES rectangle over two separate periods, 1997 to 2000 and 2001 (see Figure 8.6.2.2.1), 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 et al 1998; Piet et al 2000; Piet et al in press). Initial runs assumed “per fishing event” mortality rates of 20%, 30%, 40% and 50%. (Figure 8.7.5.1.1).

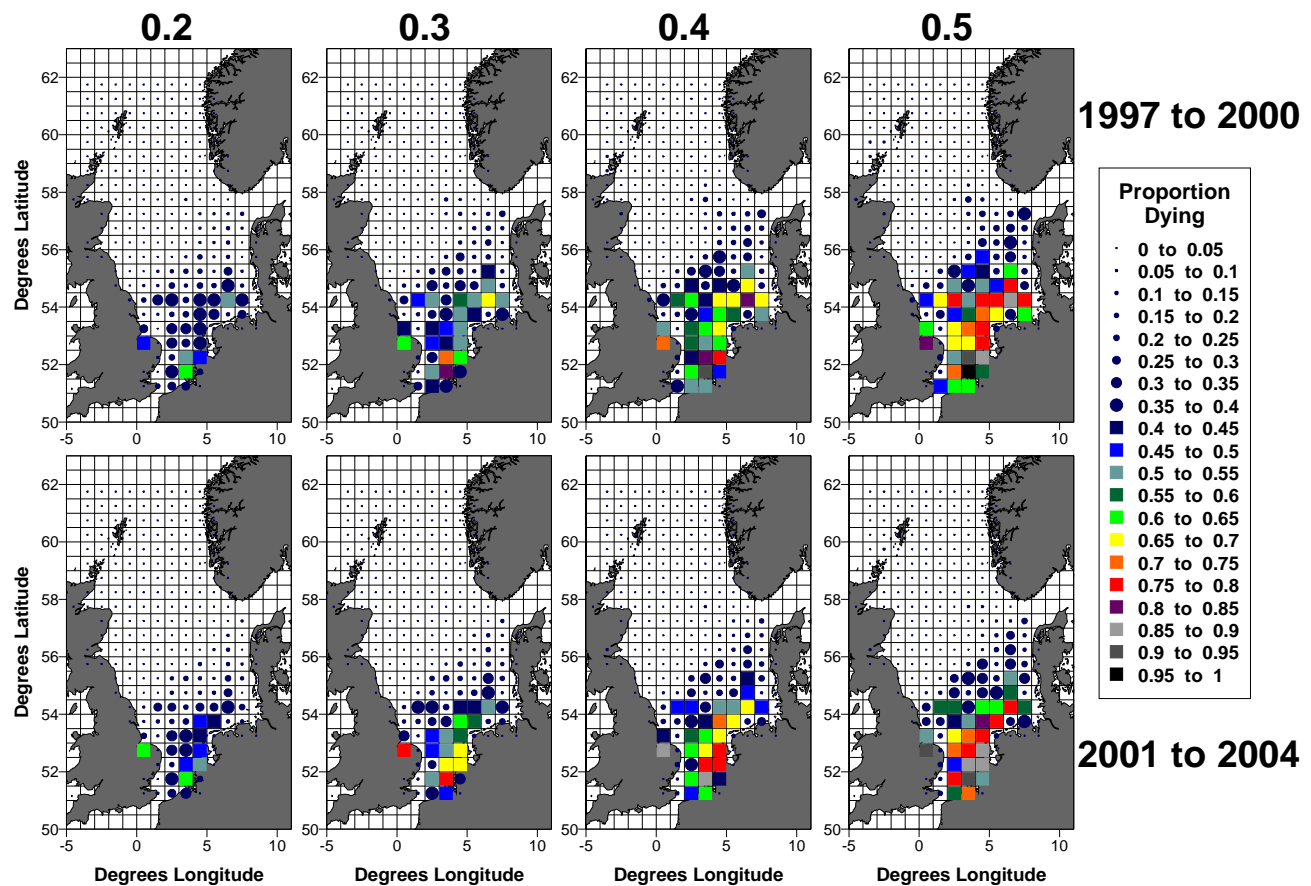


Figure 8.7.5.1.1. Modelled impact of beam trawling on benthic invertebrates. Maps show total modelled annual mortality given the distribution of beam trawl fishing activity and “per fishing event” mortality rates of 20%, 30%, 40%, and 50%.

8.7.5.2. *Otter trawling directed at fish*

We first examine otter trawling directed at fish. The model was run using annual average hours-fishing per ICES rectangle over two periods; 1997 to 2000 and 2001 to 2004 (see Figure 8.6.2.2.2). 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 8.7.5.2.1).

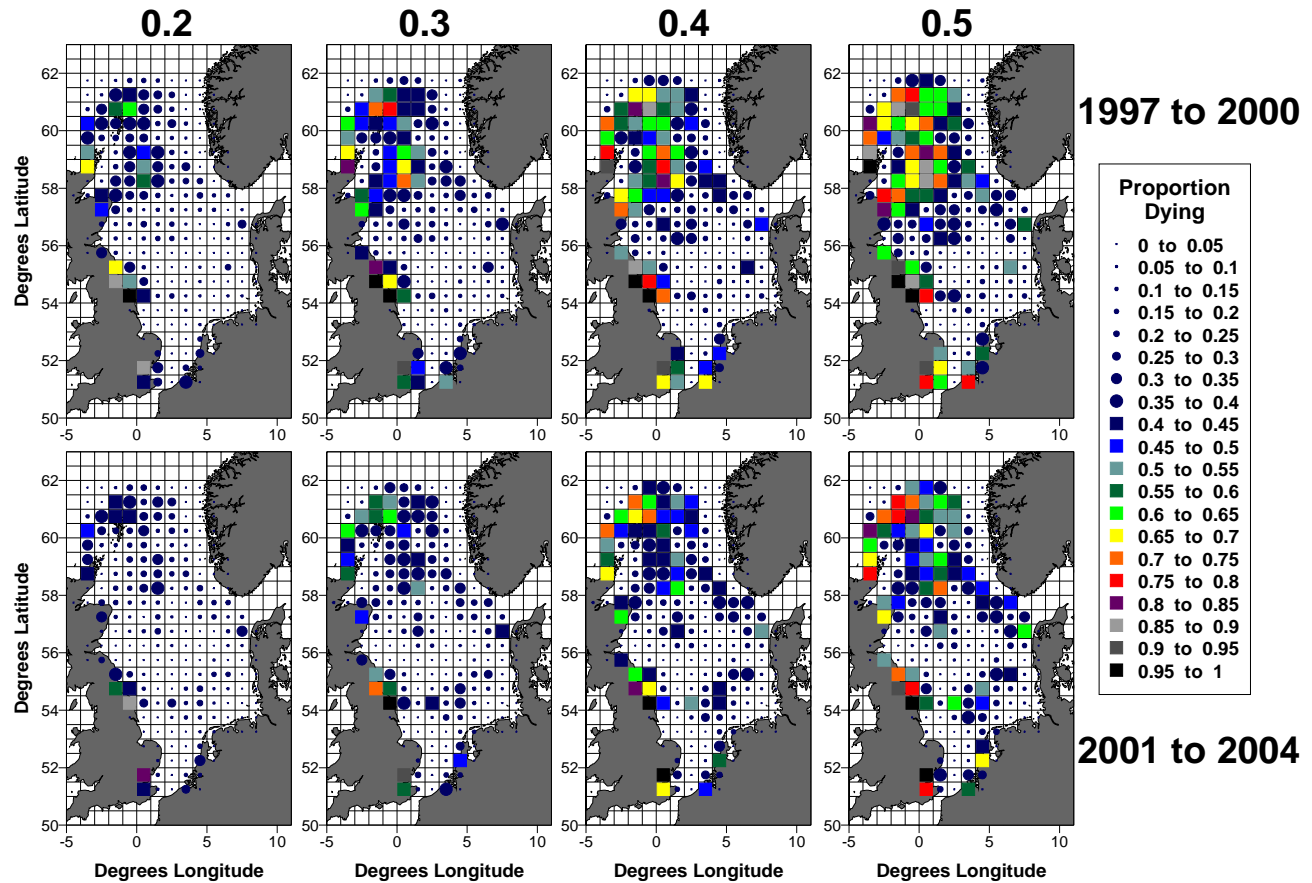


Figure 8.7.5.2.1. Modelled impact of otter trawling targeting fish on benthic invertebrates. Maps show total modelled annual mortality given the distribution of otter trawl (targeting fish) fishing activity and “per fishing event” mortality rates of 20%, 30%, 40%, and 50%.

8.7.5.3. Otter trawling directed at *Nephrops*

We now examine otter trawling directed at *Nephrops*. The model was run using the annual average hours-fishing per ICES rectangle over two periods; 1997 to 2000 and 2001 to 2004 (see Figure 8.6.2.2.3). 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 8.7.5.3.1).

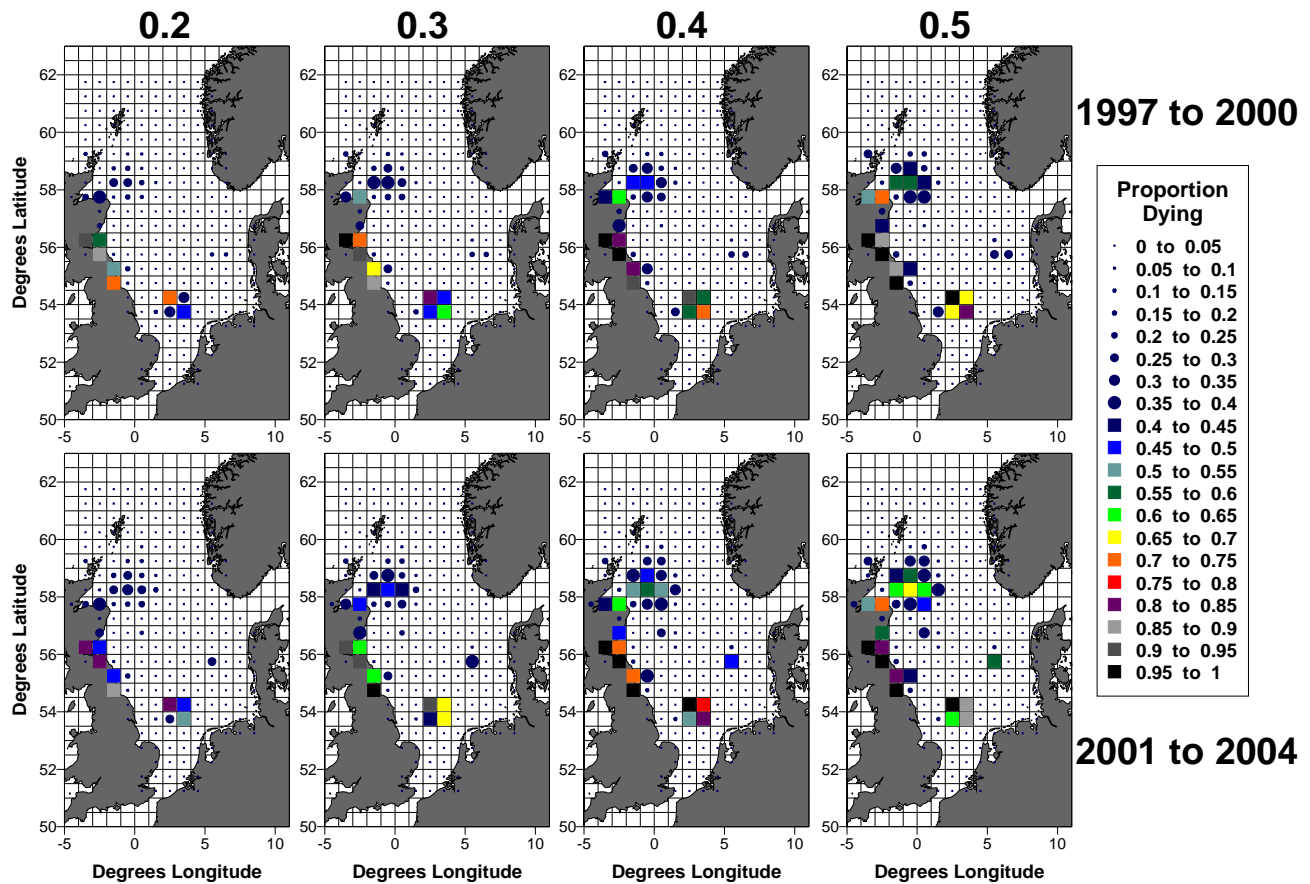


Figure 8.7.5.3.1. Modelled impact of otter trawling targeting Nephrops on benthic invertebrates. Maps show total modelled annual mortality given the distribution of otter trawl (targeting Nephrops) fishing activity and “per fishing event” mortality rates of 20%, 30%, 40%, and 50%.

8.7.5.4. 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 two periods; 1997 to 2000 and 2001 to 2004 (see Figure 8.6.2.2.4). 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 2.43km² (Figure 8.7.5.4.1). Initial runs assumed “per fishing event” mortality rates of 20%, 30%, 40% and 50%. (Figure 8.7.5.4.2).

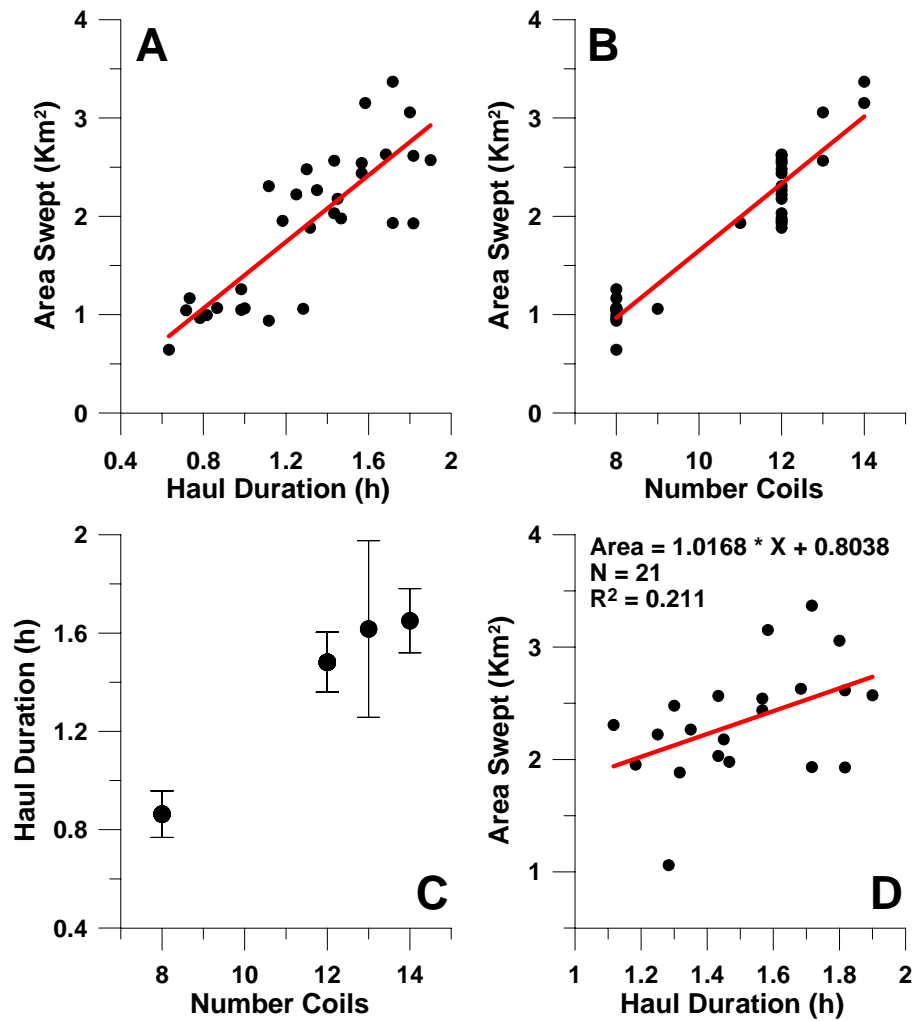


Figure 8.7.5.4.1. Analysis of Seine gear fishing parameters. A. The area swept is positively correlated with tow duration ($R^2=0.684$, $N=31$). B. However, the number of coils laid has a much closer relationship on the area swept ($R^2=0.901$, $N=31$), and the relationship shown in A is strongly influenced by this. C. Tows that laid only 8 coils were all less than 1.5h in duration. The modern day Seine gear tow duration of 1.6h therefore involves tows where 12 or more coils are laid. D. The relationship between area swept and tow duration for tows where 12 or more coils were laid is weaker, but this provides an average swept area estimate of 2.43Km² for tows of on average 1.6h duration.

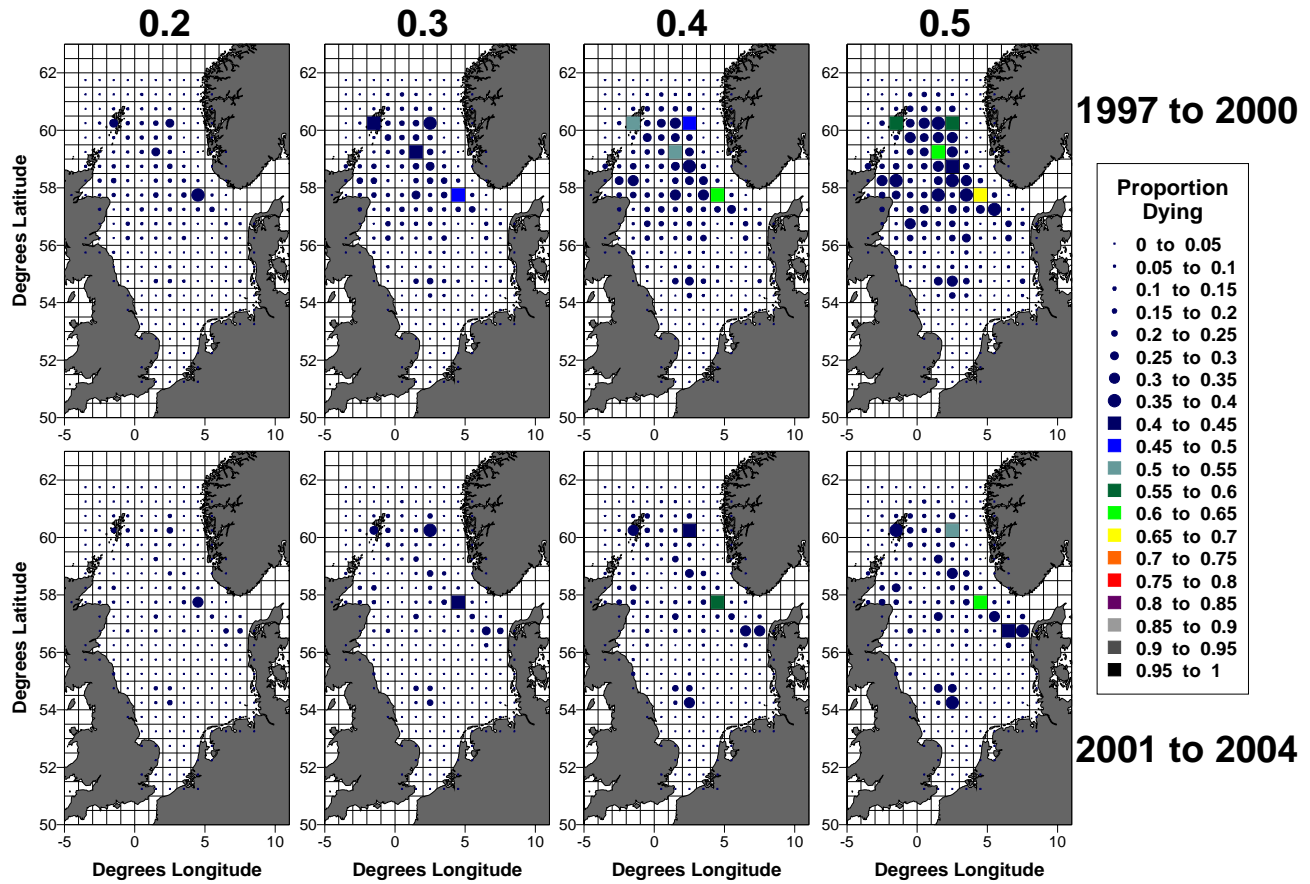


Figure 8.7.5.4.2. Modelled impact of Seine gears on benthic invertebrates. Maps show total modelled annual mortality given the distribution of Seine gear fishing activity and “per fishing event” mortality rates of 20%, 30%, 40%, and 50%.

8.7.5.5. *Modelling the Combined Impact of all Fishing*

Our 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 (Figure 8.7.5.5.1). Essentially this provided a “baseline” estimate of benthic invertebrate mortality determine for a universal generic benthic invertebrate community. Estimating the total impact of fishing for all gears combined is not a case of simply adding the mortalities estimated for each gear independently. Once again, where two or more gears operate within a single ICES rectangle, the mortality caused by each gear interacts with the mortality caused by each of the other gears. In equation 8.7.2.9, M_{RECT} is the total mortality caused in any given ICES rectangle by a specific gear. We therefore now further subscript this to M_{RECT_g} . Then the total benthic mortality caused by the operation of all four main gear categories in each ICES rectangle is $M_{RECT_{tot}}$, given by:

$$M_{RECT_{tot}} = 1 - e^{\sum_{g=1}^4 -Ln(1-M_{RECT_g})} \quad 8.7.4.5.1$$

The combined impact of each of the gear categories shown in Figure 8.7.5.5.1 is shown as a composite fishing disturbance mortality in Figure 8.7.5.5.2.

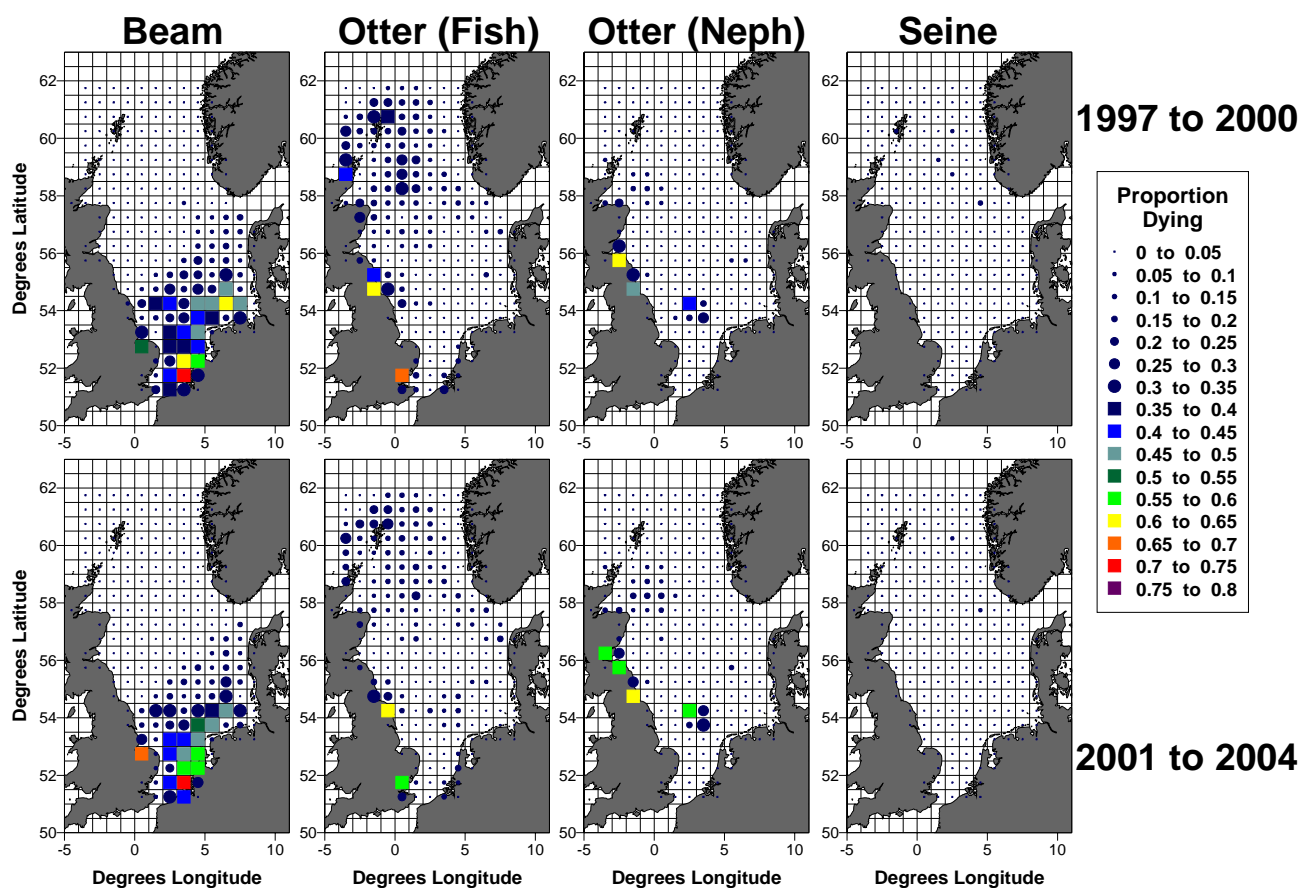


Figure 8.7.5.1. Modelled impact of four major demersal fishing categories on the benthic community of the North Sea. Maps show total modelled annual mortality given the distribution of average annual beam trawl, otter trawl targeting fish, otter trawl targeting Nephrops, and Seine gear fishing activity in two periods, 1997 to 2000 and 2001 to 2004. Generic mortality rates “per fishing event” assumed for each gear type are indicated.

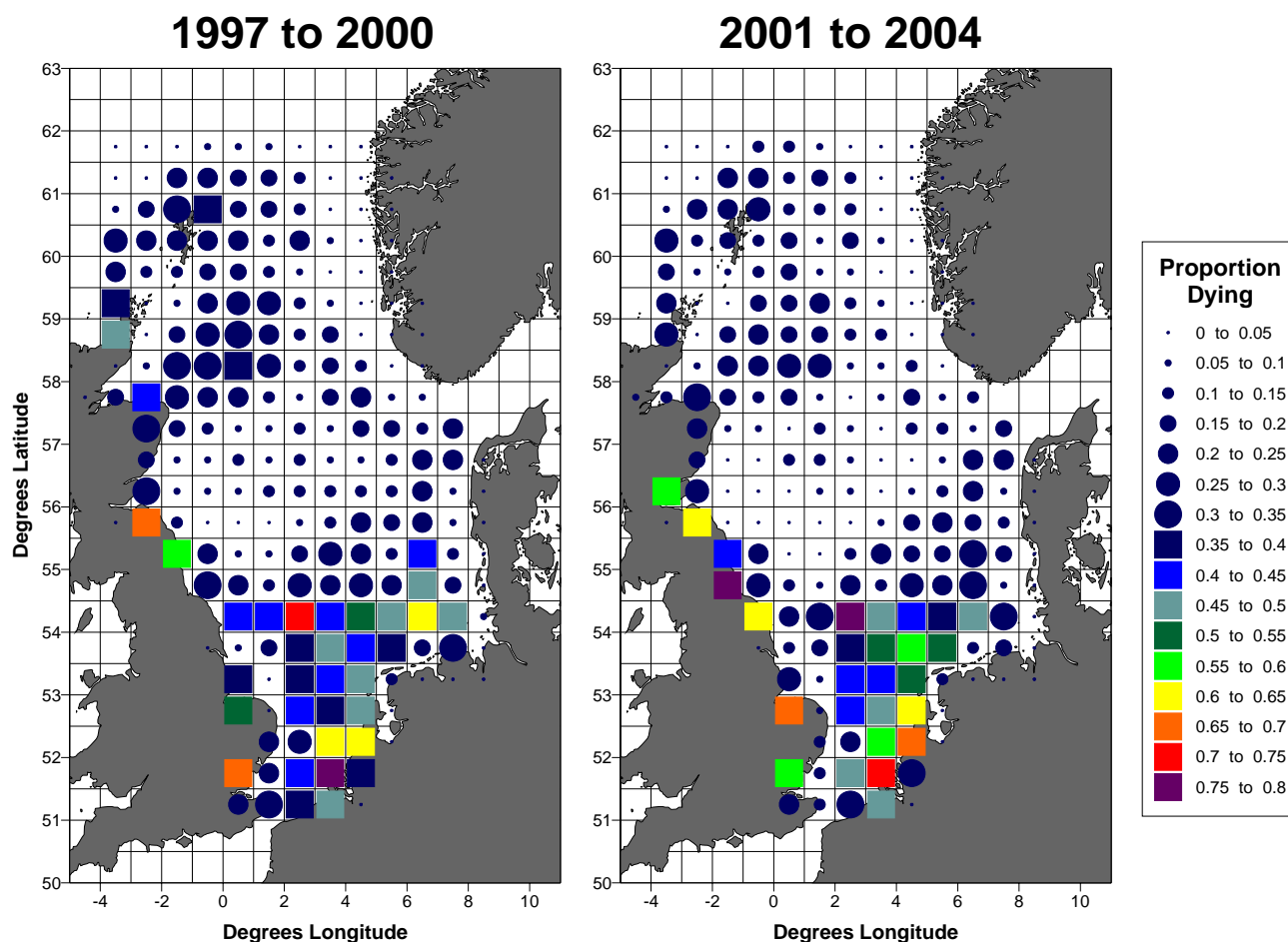


Figure 8.7.5.5.2. Modelled impact of four major demersal fishing categories on the benthic community of the North Sea. Maps show total modelled annual mortality of all four gears combined, given the average annual distribution of beam trawl, otter trawl targeting fish, otter trawl targeting Nephrops and Seine gear fishing activity in two periods, 1997 to 2000 and 2001 to 2004, and assuming generic “per fishing event” mortality rates of 25%, 10%, 10%, and 5% respectively for each gear type.

8.7.5.6. *Modelling the impact of fishing on actual epibenthic communities*

Our analyses to determine spatial variation in the abundance and distribution of epibenthic fauna is described in Section 10.3.2, with examples of the distribution of the 12 most abundant species, by biomass, presented in Figure 10.3.2.2. To model the impact of fishing on actual epibenthic communities, the biomass of each species in each rectangle was multiplied by our estimates of the proportion removed or killed by a single passage of each of the four main fishing gear categories to produce estimates of the biomass of each species destroyed by each type of fishing event. For each rectangle, summing the total biomass across all species, both before and after a single event of each type of fishing activity, then dividing the “after event” biomass total by the “before event” total provided estimates of the “community average” mortality inflicted by each type of fishing gear in each rectangle. These ICES rectangle mortality rates were then input into the benthic mortality model and the model run using the average annual effort (2001 to 2004) for each gear in each rectangle to produce estimates of spatial variation in the annual benthic invertebrate mortality caused by each gear type (Figure 8.7.5.6.1). Finally the model combines the four sets of mortality data to give the required estimates of spatial variation in the annual impact of all fishing activity on

the epibenthic invertebrate community of the North Sea. These estimates, which take account of spatial variation in the species composition of the epibenthic community, are compared in Figure 8.7.5.6.2 with a map generated using the “generic” benthic community, as described in Section 8.7.4.5.

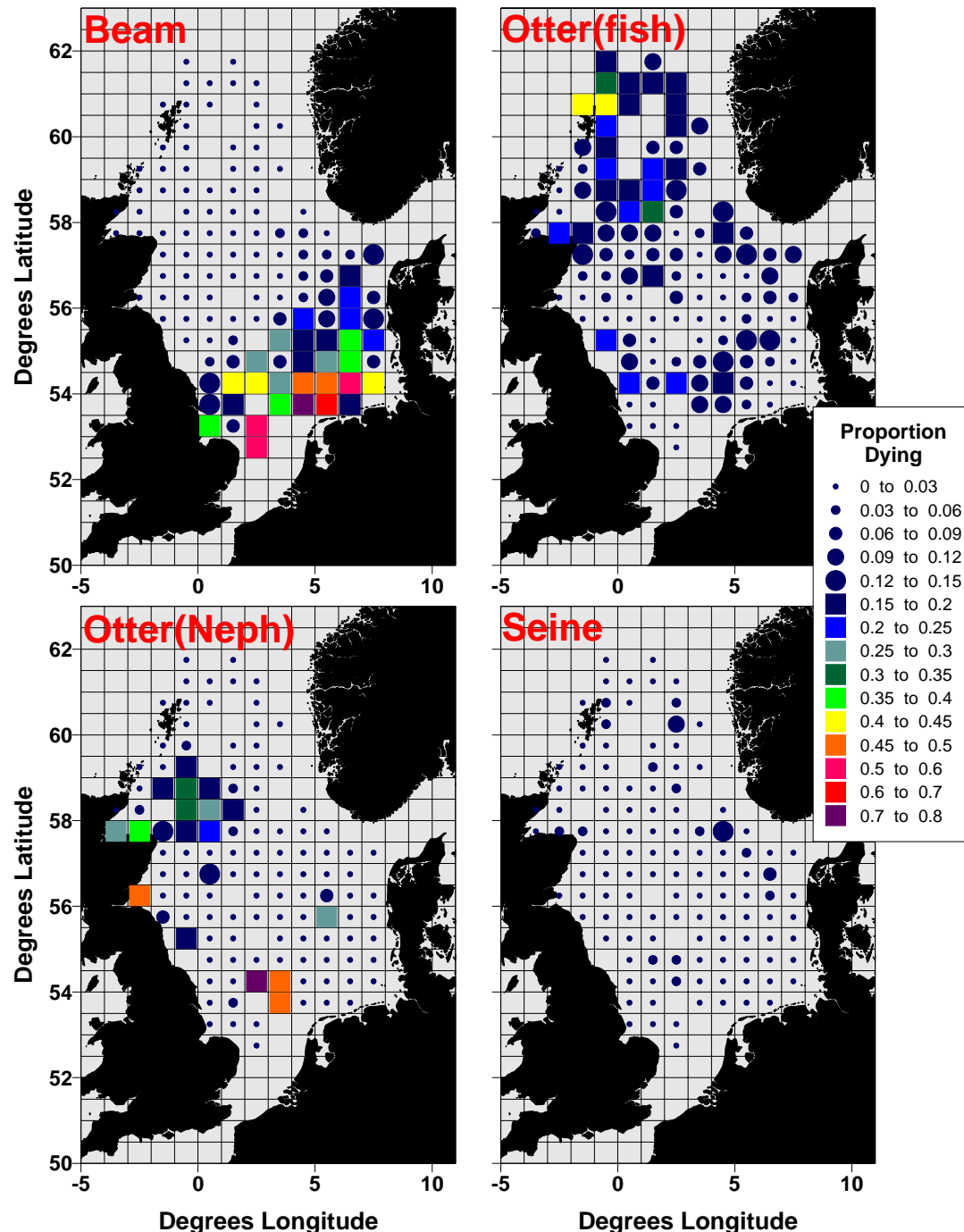


Figure 8.7.5.6.1. Modelled impact of four major demersal fishing categories on the benthic community of the North Sea. Maps show total modelled annual mortality given the distribution of average annual beam trawl, otter trawl targeting fish, otter trawl targeting Nephrops, and Seine gear fishing activity between 2001 to 2004 and using community averaged mortality rates dependent on the species composition in each rectangle.

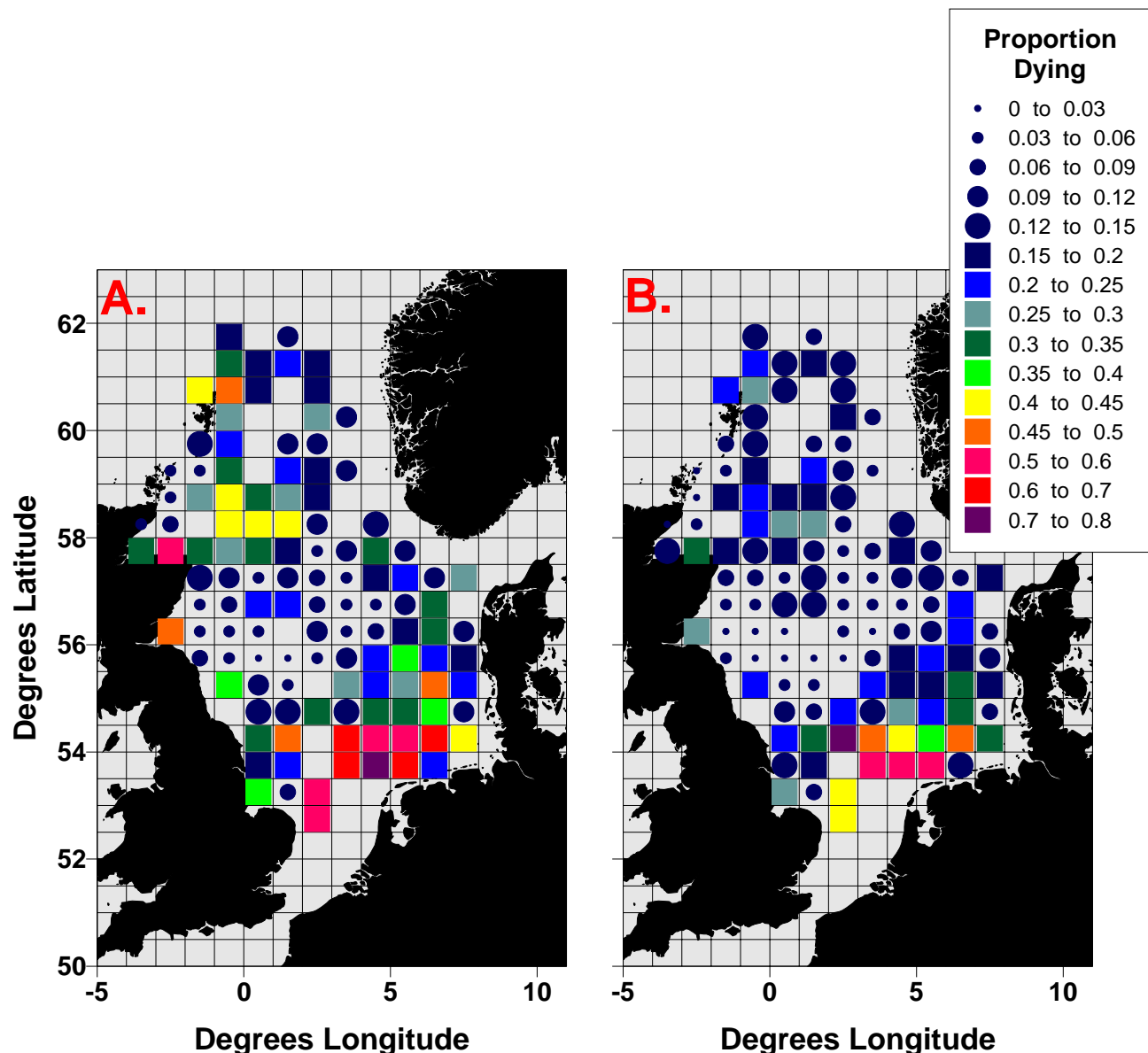


Figure 8.7.5.6.2. Modelled impact of four major demersal fishing categories on the benthic community of the North Sea. Maps show total modelled annual mortality of all four gears combined, given the average annual distribution of beam trawl, otter trawl targeting fish, otter trawl targeting Nephrops and Seine gear fishing activity between 2001 to 2004, and assuming community averaged “per fishing event” mortality rates for each gear type that were dependent on species composition in each rectangle (A), or assuming generic “per fishing event” mortality rates of 25% (beam trawl), 10% (both otter trawls), and 5% (seine net) across all rectangles (B).

8.8. Modelling the mortality of fish

8.8.1. Introduction

Piet et al., (2006) evaluated some potential pressure indicators and concluded that annual fishing mortality is the best pressure indicator because this is directly linked to state as well as the only level at which effects of technical or spatial management measures (e.g. mesh size regulations or MPAs)

can be identified. Piet et al. (2006) also stated that the pressure expressed as annual fishing mortality of one or two commercial species will not be representative for the whole community. Therefore, if management objectives are set for community level indicators such as mean weight, mean maximum length, or biodiversity (Piet & Jennings, 2005), the community mortality will need to be determined as an integral of all the population mortalities that make up the community. In theoretical ecology Huston's dynamic equilibrium model (Huston, 1994) considers species diversity to depend on productivity and disturbance. Here disturbance is the mortality caused by particular perturbation scenarios to the constituent populations that make up the communities in question. In theoretical ecology terms therefore, disturbance to a community equates to the mortality caused by the "disturbing" events (chapter 7. In attempting to model the effects of fishing activity (the "disturbing" events) on fish and benthic invertebrate communities, it is the mortality caused by the fishing activity that needs to be input as the "explanatory variable", not the measure of fishing activity (e.g. hours effort per unit area per unit time).

The mortality of a wide range of ecosystem components can be calculated following variations on the swept-area method introduced simultaneously by Pope et al. (2000) for fish and (Piet *et al.*, 2000) for benthos. This method essentially combines information on fishing effort (Jennings *et al.*, 1999; this Chapter) with data on the distribution and abundance of the biota that are often available from stock assessments (ICES ACFM) and surveys (Künitzer *et al.*, 1992; Knijn *et al.*, 1993) assuming that the gear catches all (i.e. catchability coefficient=1) or part of the fish in its track (see Chapter 9). We followed this approach to some extent but modified it in that we used true abundance estimates of fish (Chapter 9), a different method of determining catchability based on the characteristics of the gear (e.g. mesh size), and based it on both otter- and beam trawl fleets with slightly different parameters that determine the swept area. As these methods required some assumptions, we also performed sensitivity analyses to show how the model output varied depending on these assumptions. Finally, we validated the model by comparing for some of the commercial species the output of the model to estimates of landings and discards from sampling programmes.

This study 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).

8.8.2. Material & methods

8.8.2.1. *Data sets*

For the purposes of this study, only data covering the period 1998 to 2004 were analysed, using the same input data as in previous sections of this chapter. Here we deviate from the practice described in Chapter 9 by using density at length estimates derived directly from the Dutch Beam Trawl Survey (DBTS) for the suite of species for which these data are used in the annual stock assessments. The result was a combined dataset based on the IBTS and DBTS for the mean abundance at length of all demersal North Sea species.

8.8.2.2. Calculations of catchability

For the raising of the DBTS data to absolute abundance, we followed the practice described in Chapter 9. The estimated “true” abundance at age of sole (*Solea vulgaris*) and plaice (*Pleuronectes platessa*) from the VPA were used to determine the catchability with which the suite of DBTS species are multiplied to get the “true” abundance per length per ICES rectangle. For this the abundance in numbers at age per haul were converted to numbers.km⁻² using the recorded bottom track multiplied by 8m (the width of the beam trawl). If only the haul duration was recorded, then the bottom track was calculated from the haul duration using a fishing speed of 4 knots. Of the N.km⁻² the geometric mean of all hauls per rectangle of species and age ($FD_{rect,s,a}$) were used for further calculations. These numbers were multiplied by the area of the ICES rectangle, thus:

$$N_{rect,s,a} = FD_{rect,s,a} * A_{rect} * pS_{rect} = FD_{rect,s,a} * S_{rect} \quad 8.8.2.2.1$$

However, the area of an ICES rectangle is not constant. While the height of each rectangle, delineated by 0.5°latitude (=30 NM or approximately 55.6 km), remains constant, rectangle width, delineated by 1.0°longitude decreases with increasing latitude. The area (km²) of each ICES rectangle (A_{rect}) is therefore given by:

$$A_{rect} = 30 * 60 * \cos(LAT_{rect}) * 1.853^2 = 6080.4342 * \cos(LAT_{rect}) \quad 8.8.2.2.2$$

where LAT_{rect} is the latitude of the rectangle mid-point. Furthermore, rectangles around the edges of the North Sea containing coastline did not consist entirely of “sea-area”. Thus, the area of each ICES rectangle was further modified by multiplying by the proportion consisting of “sea-area” (pS_{rect}) resulting in the “sea-area” of each rectangle (S_{rect}).

The abundances at age per rectangle were summed by sub-area as described in Chapter 9. Then multiplied by the raising factor (RF) calculated as the total surface of the sub-area divided by the surface of the rectangles fished by the DBTS in that sub-area. These raised abundances were summed over the sub-areas and then divided by the mean abundances from the VPA in the 3rd quarter to calculate the catchability (q). As the DBTS takes place in the 3rd quarter the VPA abundance in the 3rd quarter ($A_{3,a,y}$) were derived from those in the 1st quarter according to:

$$A_{3,a,y} = (A_{1,a,y} + A_{1,(a+1),(y+1)})/2 \quad 8.8.2.2.3$$

Because the survey data are length based, estimates of q at age for the two species needed to be converted to q at each 1 cm length class, above sixty centimeter catchability was assumed constant. For each 1 cm length class the q of the age class that contributes most to this length group was used. The linking of length and age classes was based on the survey-based age-length keys of each species. Age 0 was not included in the analyses, thus all fish below 12-cm were not included.

We followed (Sparholt, 1990) attributing plaice and sole to respectively groups 6 and 7 and only considered those species belonging to these groups. We assumed that a non-target fish of equal size as the commercial species representative of that group had an equal catchability. The catchabilities were multiplied by the geometric mean abundance at length per rectangle to get the absolute abundances per length of all species per rectangle. The results for sole and plaice from the DBTS are compared to those obtained by the similar analysis of the IBTS data set in Chapter 9. The DBTS had a smaller coverage of the North Sea compared to the IBTS, therefore only those rectangles of the IBTS were replaced where at least five DBTS hauls were taken during these 7 years. The numbers in the new dataset were converted to biomass by:

$$B_{rect,s,l} = N_{rect,s,l} * L^3 \quad 8.8.2.2.4$$

where: $B_{\text{rect},s,l}$ is the biomass (kg) per species per length per rectangle; $N_{\text{rect},s,l}$ is the numbers per species per length per rectangle; and L is length (m). These Biomasses were used as input for the model.

8.8.2.3. Trawling frequency

In Piet et al. (2000) the frequency with which an area was trawled is considered to be a better measure of fishing impact than conventional effort measures such as days-at-sea or hours fished. For fishing effort (hours fished) we used the modelled data of Greenstreet (in prep) on the international otter and beam trawl effort for the period 1998-2004. Trawling frequency (F_t) was calculated as:

$$F_t = \text{Eff}_w * T_F * V * S_{\text{rect}}^{-1} \quad 8.8.2.3.1$$

Where: F_t is the frequency trawled; Eff_w is the effective width (m); T_F is the time fished (s); V is the velocity (m/s); and S_{rect} is the “sea area” of ICES rectangle (m^2)

The parameter values used for the otter trawl were those used in previous sections of this chapter. The parameter values used for the beam trawl were obtained from Piet et al. (2006) (table 8.8.2.3.1). The effort in hours for the beam trawl was multiplied by the ratio between the hours fished by large vessels (>221 Kw) and small vessels ‘eurocutters’ (<221 Kw) per rectangle.

Gear	Kw	Speed (m/s)	Eff_w (m)
Beam trawl	<221	7.78	2*4
Beam trawl	>221	12.41	2*12
Otter trawl	all	5.00	87

Table 8.8.2.3.1. Fishing effort parameter values.

8.8.2.4. 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 (Engås & Godø, 1989; Weinberg *et al.*, 2002) we developed a conceptual framework in which catch efficiency is determined by four factors:

1. Positioning in the water column
2. Herding
3. Escape below footrope
4. Retention in the net

Some of these factors are discussed in more detail below. Numerous other factors may affect catch efficiency. For example vessel noise (Dickson, 1993), visibility, fishing speed, density-dependent catchability, diel variation and mesh shape (Robertson *et al.*, 1988; Wardle, 1988; Godø *et al.*, 1999; Weinberg *et al.*, 2002; Benoit & Swain, 2003). 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 were no quantitative data we assumed that 80% of the

roundfish were 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 was assumed to be only 30% for the beam trawl. Flatfish were 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; Engås & Godø, 1989; Dickson, 1993; 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 & Godø, 1989). However, for simplicity herding can be assumed independent of fish length (Ramm & Xiao, 1995). We assume a correction factor of 0.75 for large roundfish (>29.5 cm) and a correction factor of 0.3 for small roundfish. No quantitative data on herding were found for flatfish. According to (Winger *et al.*, 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 of 0.3 for flatfish. 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 & 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 & 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; Beek *et al.*, 1983; Robertson *et al.*, 1988; Reeves *et al.*, 1992; 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 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):

$$S_L = \{3^{(L50 - (L + \Delta L/2))/(L50 - L25)} + 1\}^{-1} \quad 8.8.2.4.1$$

where: S_L is the proportion of the population of length L and class width ΔL that is retained; $L50$ is the length of which 50 percent of the population entering the net is retained (cm); and $L25$ is 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) (Table 8.8.2.4.1):

$$L50 = SF * M \quad 8.8.2.4.2$$

$$L25 = L50 - (SR/2) \quad 8.8.2.4.3$$

where: SF is the selection Factor; M is the mesh size (cm); and SR is the election range (cm). The mesh size used for the beam trawl is 8 cm and 10 cm for the otter trawl; this is based on the mesh size used most by the commercial fleet. As sufficient quantitative information to determine cod-end selectivity is only available for some commercial species (MacLennan, 1992) we determined selectivity parameters for roundfish and flatfish and applied those to the non-target species.

Species	SF_BT	SR_BT	SF_OT	SR_OT
Cod	3	7.4	3	8.1
Haddock	3.1	6.6	3.1	6.6
Whiting	3.5	7	3.2	7.3
Saithe	4.2	5.6	4.3	5.7
Dab	2.2	4.1	2.5	1.9
Plaice	2.2	4	2.5	1.6
Sole	3.2	3.8	3.2	3.8
Roundfish	3.5	6.7	3.4	6.9
Flatfish	2.2	4	2.5	1.7

Table 8.8.2.4.1. 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. The parameters are within the range found by (Wileman, 1991; Wileman *et al.*, 1996).

The values for the positioning, herding and footrope (Small/Large fish) factor were assumed constant (Table 8.8.2.4.2). These factors were multiplied to result in a final efficiency factor. Thus a beam trawl is more selective than an otter trawl for flatfish (0.95 versus 0.14 for small and 0.24 for large flatfish) and less selective for large roundfish (0.28 versus 0.41 for roundfish) (table 8.8.2.4.2).

Gear	Fish type	Fish length (cm)	Factor			
			Positioning	Herding	Footrope	Overall
BT	DR	all	0.28	1	1	0.28
BT	DF	all	0.95	1	1	0.95
OT	DR	<29.5	0.57	0.3	0.95	0.16
OT	DR	>29.5	0.57	0.75	0.95	0.41
OT	DF	<25	0.95	0.3	0.5	0.14
OT	DF	>25	0.95	0.3	0.85	0.24

Table 8.8.2.4.2. 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). The factor is dependent on both fish- and mesh-size. The overall factor was calculated by multiplying the positioning, herding and footrope factor.

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.

8.8.2.5. On-board selection

Fishing gears catch individuals of both commercial and non-target species (Heessen & 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. On-board selection

determines which part of the fish caught is actually landed, the remainder being discarded. (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.

8.8.2.6. *Mortality*

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

$$C_g = (1 - PR_g) * F_t \quad 8.8.2.6.1$$

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

$$M = B * (1 - C_b * C_o) \quad 8.8.2.6.2$$

Where B is the biomass 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 percentage discarded is calculated as the biomass discarded/biomass caught. The percentage mortality is the biomass caught/biomass present.

8.8.2.7. *Timestep*

The model takes in account the redistribution of fish after passing of a trawl. Introducing the effort in 12 pieces and between every step redistributing the fish that survived in the same way as that they were distributed before fishing took place does this. In this way, the catches are slightly higher than when all the effort takes place in one time. The idea behind this is that it makes it possible to introduce seasonal effort and fish data, if they were to become available.

8.8.2.8. *Validation*

The outcome of the model was estimated landings of the commercial species and estimated discards of all species. For the validation of the outcome, we used the modelled total landings data (section 8.6.1) on the mean international otter and beam trawl landings for the period 1998-2004 and North Sea discard data from <http://stecf.jrc.cec.eu.int/meetings/sgrn/0606/reportandannex.pdf>

8.8.2.9. *Sensitivity analysis*

A sensitivity analysis was done on the overall factor of catch efficiency. For this the results of the factors described above are compared to the overall factors as described in MAFCONS Deliverable 12 and a third set of factors and combinations of these factors.

Gear	Fish type	Fish length (cm)	Factor		
			Overall	MAFCONS deliverable 12	Third set
BT	DR	all	0.28	0.3	0.45
BT	DF	all	0.95	1	0.8
OT	DR	<29.5	0.16	0.65	0.4
OT	DR	>29.5	0.41	0.65	0.2
OT	DF	<25	0.14	0.12	0.3
OT	DF	>25	0.24	0.21	0.4

Table 8.8.2.9.1. Factors used in the sensitivity analysis of 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). The factor is dependent on both fish- and mesh-size. Overall is the set as used in this study, MAFCONS deliverable 12 is de set as described there and the third set is a set of chosen values to be different of the other sets.

8.8.3. Results

8.8.3.1. *Comparison DBTS set and VPA.*

The number of sole and plaice after raising by length class with the catchability should result in data close to the biomass in the VPA. In Figure 8.8.3.1.1 the result of the comparison is shown.

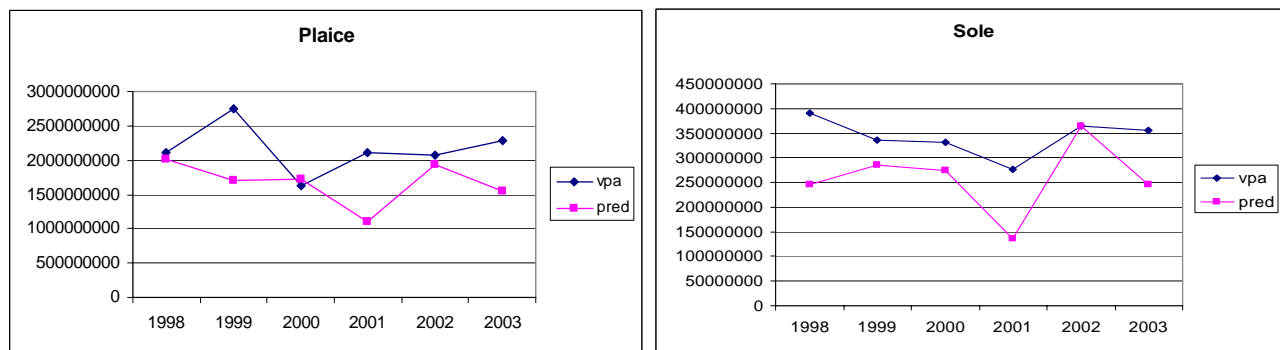


Figure 8.8.3.1.1. The estimated number of plaice and sole by year raised by length multiplied by the RF (raising factor for sub-area) summed over the North Sea (pred) compared to the VPA data (vpa).

8.8.3.2. *Comparison IBTS and DBTS sets.*

Plaice was calculated in the same way for both sets and resulted in comparable estimated numbers per length. This study estimated the number of sole in the same way, this resulted in higher numbers for most length classes and as seen in the comparison with the VPA data, these numbers were even lower than the numbers in the VPA. Showing that the IBTS based estimates are too low for the species in Sparholt group 7 (Figure 8.8.3.2.1).

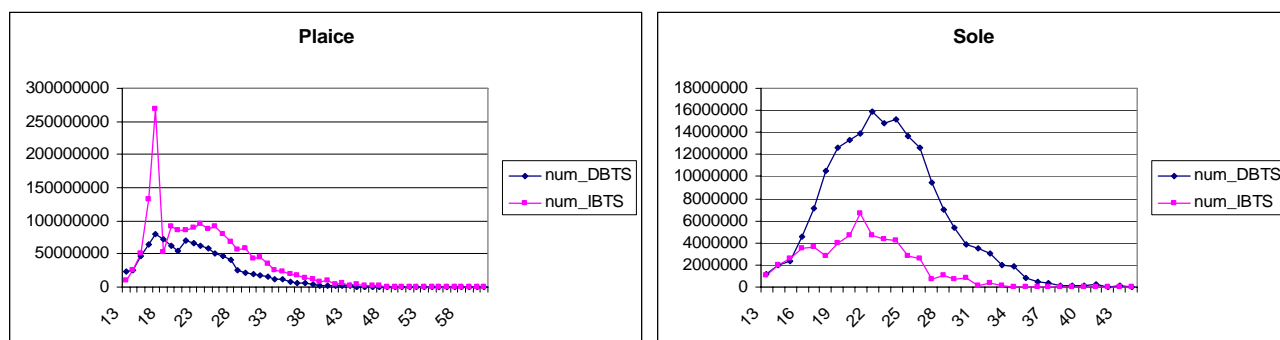


Figure 8.8.3.2.1. Comparison between the IBTS data and the DBTS by which it was replaced.

8.8.3.3. Validation of the model

The model-based estimates of landings in biomass were compared with the actual total landings. The total estimated landings were 0.6 up to 1.47 of the actual landings, except saithe that is much more overestimated. Looking at the otter trawl that is primary used to catch the round fish, and then it shows that it slightly overestimated the catch of roundfish and underestimated the catch of flatfish. The beam trawl was in the right direction considering the flatfish and overestimated landings of the roundfish. Nevertheless, sole is still underestimated. The discards were presented as percentages of the total catch and were thus influenced by the landings as by the discards. The results for the beam trawl indicated that as well as the landings the discards of flatfish were in the right direction. The otter trawl underestimated the landings as well as the discards. For the roundfish the otter trawl underestimated the discards, but the beam trawl overestimated them for cod and haddock.

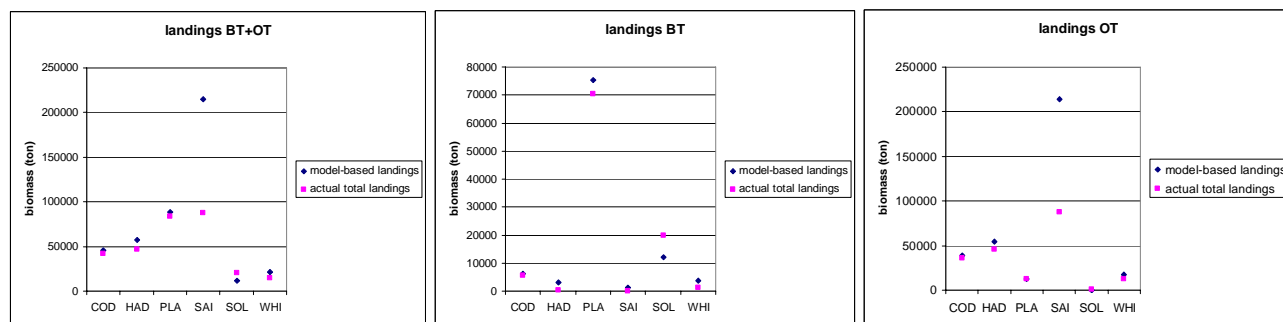


Figure 8.8.3.3.1. The model-based estimates of landings (t) compared to the actual landings (t).

species	estimated landing total(ton)	estimated landings OT(ton)	estimated landings BT(ton)	Landings total(ton)	Landings OT(ton)	Landings BT(ton)	fraction total	fraction OT	fraction BT
COD	45643	39362	6282	41747	36096	5651	1.09	1.09	1.11
HAD	57153	54202	2952	46386	46085	301	1.23	1.18	9.80
WHI	21212	17513	3699	14382	13026	1357	1.47	1.34	2.73
SAI	215424	214217	1208	87183	87170	13	2.47	2.46	93.87
PLA	88387	12885	75501	83425	13064	70361	1.06	0.99	1.07
SOL	12121	148	11973	20237	545	19692	0.60	0.27	0.61

Table 8.8.3.3.1: The model-based estimates of landings compared to the actual landings and the fraction (estimated/actual).

species	Estimated OT	Estimated BT	STECF OT	STECF BT
COD	11.7	28.7	21.3	7.2
HAD	14.9	52.4	22.7	20
WHI	8.8	55.8	40.8	72.8
PLA	11.7	49.6	17.4	49.5
SOL	2.6	12.4	9.4	13

Table 8.8.3.3.2. The model-based estimates of discards compared to the actual discards. It are the percentages of the total catch (discards/(discards+landings) *100).

8.8.3.4. Sensitivity analysis

The overall factor of catch efficiency is now chosen in such a way that the factors for herding, positioning and footrope are within the range of reported values or are realistic in our opinion. This was already the case for the factors as reported in MAFCONS deliverable 12, however the factors used were tuned with the landings and discards. The total model-based mortality changes of course between the three overall factors used. The overall factors used now result in lower total mortalities than when the factors of MAFCONS deliverable 12 would have been used (Figure 8.8.3.4.1). If we only change one of the factors in the overall set into one of the factors of the other sets, thus for example only changing the factor for flatfish in the beam trawl, this results in different total mortalities (Figure 8.8.3.4.2). It is shown that changing the factor of beam trawl on flatfish has the most effect on the overall mortality of flatfish, changing the factor of otter trawl on large round fish has the most effect on the roundfish. Of course has changing a factor of roundfish no effect on flatfish and the other way around. Changes in one of the gears effects the catches of the same species of the other gear.

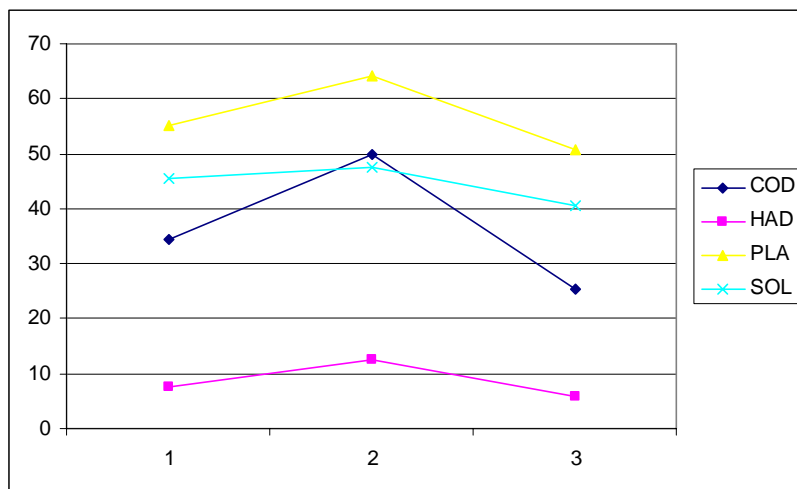


Figure 8.8.3.4.1. The effect on the overall mortality of cod, haddock, plaice and sole when using the three factors for catch efficiency. 1: Overall, 2: MAFCONS Deliverable 12, 3: Third set.

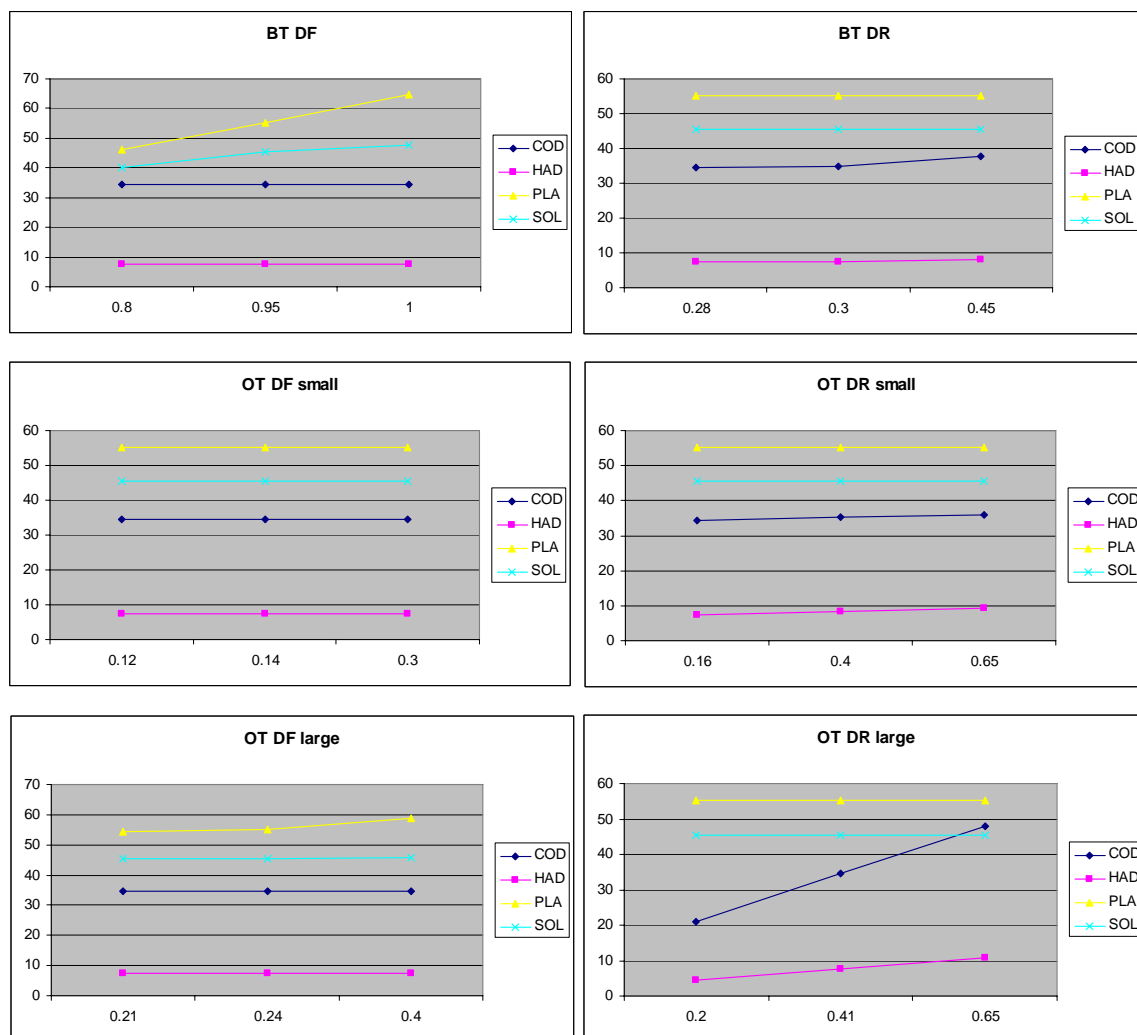


Figure 8.8.3.4.2. The effect of changing one of the factors on the commercial species cod, haddock, plaice and sole.

8.8.3.5. Mortality

The total model-based mortality in biomass of all demersal North Sea species caused by otter and beam trawling ranges from 0 to 98% (Table 8.8.3.5.1). To show the mortality on the different length class, species are grouped. The commercial group being (cod, sole, plaice, saithe, whiting, haddock); the other species are grouped in non-target flatfish, non-target round fish or elasmobranchs. It is shown that of the larger length classes the mortality is up to 40 to 60% (Figure 8.8.3.5.2.), while the in biomass the top is in the smaller classes (Figure 8.8.3.5.1.).

Species	Biomass (ton)	Survive (ton)	Mortality (ton)	Mortality (%)
AGONUS CATAPHRACTUS	1408.328	1406.561	1.766	0.13
AMBLYRAJA RADIATA	79730.403	51774.251	27956.152	35.06
ANARHICHAS LUPUS	7646.675	4313.232	3333.443	43.59
ANARHICHAS MINOR	0.087	0.087	0.000	0.11
ANGUILLA ANGUILLA	7801.509	3978.269	3823.241	49.01
ARNOGLOSSUS IMPERIALIS	7.319	7.198	0.121	1.65

ARNOGLOSSUS LATERNA	1410.013	1215.184	194.829	13.82
ASPIRIGLA CUCULUS	665.698	622.377	43.321	6.51
BROSME BROSME	2395.827	1336.229	1059.597	44.23
BUGLOSSIDIUM LUTEUM	6186.435	6077.756	108.679	1.76
CALLIONYMUS LYRA	11596.988	11461.016	135.972	1.17
CALLIONYMUS MACULATUS	515.929	511.923	4.006	0.78
CALLIONYMUS RETICULATUS	47.855	47.846	0.009	0.02
CAPROS APER	2.317	2.317	0.000	0.02
CHIMAERA MONSTROSA	33.498	22.396	11.103	33.14
CILIATA MUSTELA	0.617	0.616	0.001	0.16
CONGER CONGER	23.678	22.057	1.621	6.85
CRYSTALLOGOBIUS LINEARIS	0.438	0.438	0.000	0.01
CYCLOPTERUS LUMPUS	123.082	96.566	26.516	21.54
DIPTURUS BATIS	3218.274	2107.273	1111.001	34.52
ECHIICHTHYS VIPERA	940.548	937.660	2.888	0.31
ECHIODON DRUMMONDI	314.164	305.117	9.048	2.88
ENCHELYOPUS CIMBRIUS	9757.768	9397.029	360.740	3.70
ENTELURUS AEQUORAEUS	709.584	485.727	223.857	31.55
ETMOPTERUS SPINAX	24.361	17.865	6.496	26.66
EUTRIGLA GURNARDUS	159459.757	140951.686	18508.071	11.61
GADUS MORHUA	146369.922	92979.955	53389.967	36.48
GAIDROPSARUS VULGARIS	241.684	236.320	5.363	2.22
GALEORHINUS GALEUS	3444.621	2668.048	776.573	22.54
GALEUS MELASTOMUS	59.177	45.842	13.335	22.53
GLYPTOCEPHALUS CYNOGLOSSUS	21390.533	15600.217	5790.316	27.07
HELICOLENUS DACTYLOPTERUS	740.755	737.494	3.261	0.44
HIPPOGLOSSOIDES PLATESSOIDES	57750.686	50138.418	7612.268	13.18
HIPPOGLOSSUS HIPPOGLOSSUS	5269.104	3590.409	1678.695	31.86
ICELUS BICORNIS	1.057	1.053	0.003	0.32
LAMPETRA FLUVIATILIS	63.591	37.455	26.137	41.10
LEPIDORHOMBUS WHIFFIAGONIS	22713.091	15232.848	7480.243	32.93
LEPTOCLINUS MACULATUS	2.283	2.278	0.005	0.22
LEUCORAJA CIRCULARIS	794.301	547.885	246.416	31.02
LEUCORAJA NAEVUS	21993.317	14154.823	7838.494	35.64
LIMANDA LIMANDA	180108.724	104487.278	75621.446	41.99
LIPARIS LIPARIS	0.192	0.192	0.000	0.01
LIPARIS MONTAGUI	0.013	0.013	0.000	0.00
LIZA AURATA	109.778	41.284	68.494	62.39
LOPHIUS BUDEGASSA	7.591	7.513	0.079	1.04
LOPHIUS PISCATORIUS	23404.790	14700.878	8703.912	37.19
LUMPENUS LAMPRETAEFORMIS	662.789	634.830	27.959	4.22
LYCODES ESMARKI	3419.157	1804.514	1614.642	47.22
LYCODES VAHLI	19.475	19.302	0.173	0.89
MELANOGRAMMUS AEGLEFINUS	884970.645	815060.863	69909.782	7.90
MERLANGIUS MERLANGUS	267229.508	239672.979	27556.529	10.31
MERLUCCIOUS MERLUCCIOUS	45679.196	27542.946	18136.250	39.70
MICROCHIRUS VARIEGATUS	168.126	161.769	6.357	3.78
MICROSTOMUS KITT	51771.619	37411.473	14360.147	27.74

MOLVA DIPTERYGIA	17.236	11.893	5.344	31.00
MOLVA MOLVA	74542.146	34490.274	40051.873	53.73
MUGIL CEPHALUS	127.022	55.379	71.643	56.40
MULLUS BARBATUS	4.915	4.530	0.385	7.83
MULLUS SURMULETUS	1137.578	1065.476	72.101	6.34
MUSTELUS ASTERIAS	11614.380	7561.759	4052.621	34.89
MUSTELUS MUSTELUS	7365.929	5210.432	2155.497	29.26
MYOXOCEPHALUS SCORPIUS	1568.018	1547.225	20.793	1.33
MYXINE GLUTINOSA	6010.761	4210.357	1800.404	29.95
PETROMYZON MARINUS	69.614	53.093	16.521	23.73
PHOLIS GUNNELLUS	23.395	23.316	0.079	0.34
PHRYNORHOMBUS NORVEGICUS	4.064	4.059	0.005	0.13
PHYCIS BLENNOIDES	65.003	41.888	23.115	35.56
PLATICHTHYS FLESUS	2301.313	455.373	1845.940	80.21
PLEURONECTES PLATESSA	225734.365	61300.995	164433.370	72.84
POLLACHIUS POLLACHIUS	3958.520	2117.975	1840.546	46.50
POLLACHIUS VIRENS	631685.697	416167.686	215518.012	34.12
POMATOSCHISTUS MICROPS	0.204	0.204	0.000	0.02
POMATOSCHISTUS MINUTUS	142.732	142.709	0.023	0.02
PSETTA MAXIMA	3728.339	763.507	2964.832	79.52
RAJA BRACHYURA	322.728	6.096	316.632	98.11
RAJA CLAVATA	3771.816	1002.560	2769.256	73.42
RAJA FULLONICA	37.446	32.834	4.612	12.32
RAJA MONTAGUI	4044.241	1120.006	2924.235	72.31
RAJA OXYRHINCHUS	85.369	66.108	19.260	22.56
SCOPHTHALMUS RHOMBUS	1376.918	114.098	1262.821	91.71
SCYLIORHINUS CANICULA	53900.967	33830.113	20070.854	37.24
SEBASTES MARINUS	419.700	144.997	274.703	65.45
SEBASTES VIVIPARUS	710.224	695.310	14.914	2.10
SOLEA LASCARIS	10.181	5.867	4.314	42.38
SOLEA VULGARIS	25173.212	11346.924	13826.288	54.92
SPONDYLIOSOMA CANTHARUS	26.924	19.835	7.090	26.33
SQUALUS ACANTHIAS	14166.055	10307.247	3858.808	27.24
SYNGNATHIDAE	22.078	18.440	3.638	16.48
SYNGNATHUS ACUS	107.833	54.899	52.934	49.09
SYNGNATHUS ROSTELLATUS	19.002	18.986	0.017	0.09
TAURULUS BUBALIS	12.383	12.345	0.037	0.30
TRACHINUS DRACO	50.566	44.556	6.010	11.89
TRIGLA LUCERNA	4340.326	3112.196	1228.130	28.30
TRIGLOPS MURRAYI	14.332	14.328	0.004	0.03
TRISOPTERUS ESMARKI	1238468.089	1237706.385	761.704	0.06
TRISOPTERUS LUSCUS	2038.941	1991.905	47.036	2.31
TRISOPTERUS MINUTUS	24154.258	24094.960	59.298	0.25
ZEUGOPTERUS PUNCTATUS	3.874	3.869	0.005	0.13
ZOARCES VIVIPARUS	13.381	13.368	0.013	0.10

Table 8.8.3.5.1. Total estimated biomass in ton of the demersal species in the North Sea and the survival after a year fishing. The total estimated mortality in tonnes and in percentage.

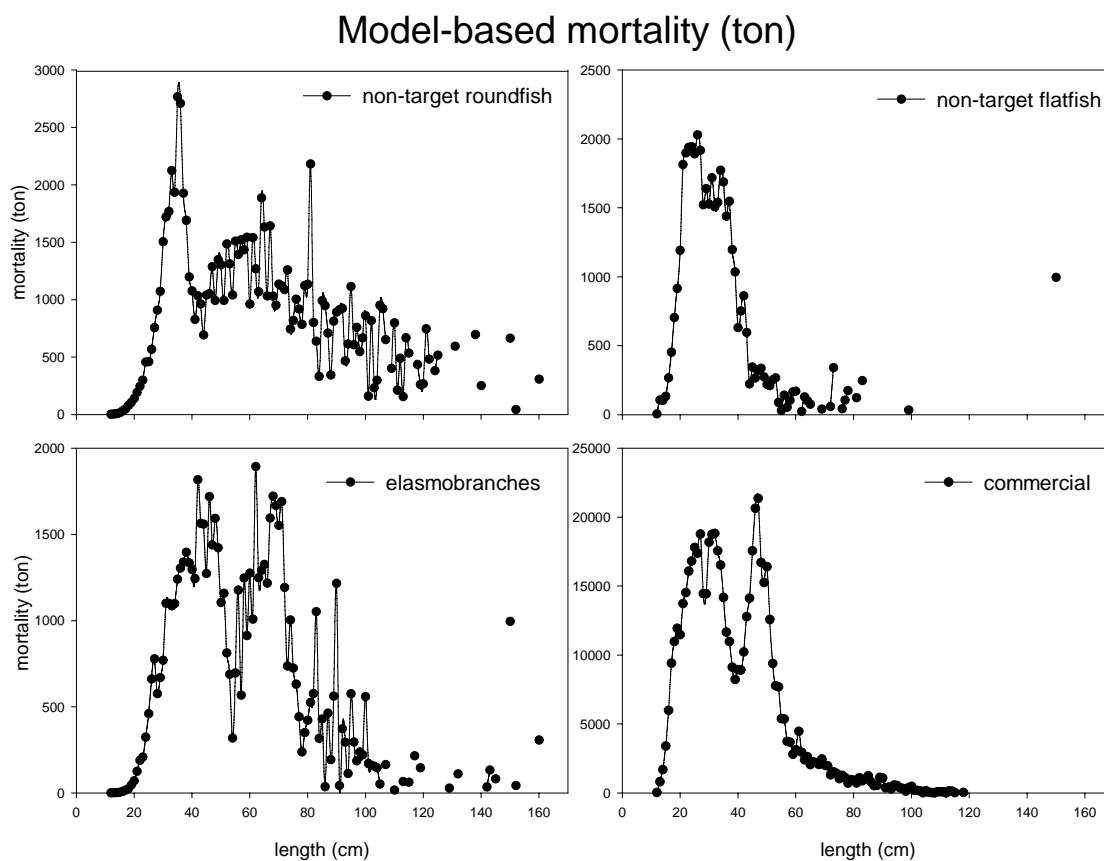


Figure 8.8.3.5.1. Model-based mortality by length class of groups of species in biomass (t).

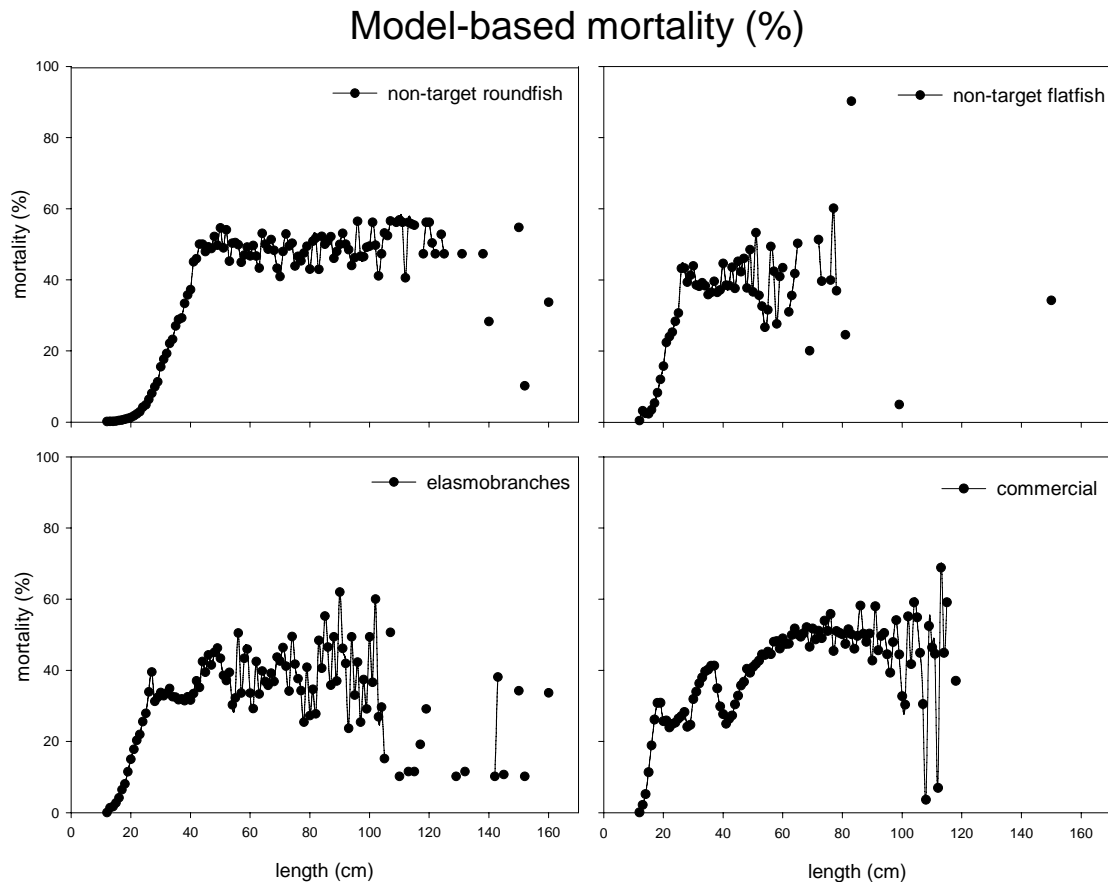


Figure 8.8.3.5.2. Model-based mortality by length class of groups of species in percentages of the total biomass of that length class.

8.8.3.6. *Distribution*

The most interesting part of this exercise is presenting the catches and mortalities of the non-commercial species and the landings and discards of the commercial species spatially. Looking at the total mortality of the total demersal community it shows distinct areas with high and low mortalities (Figure 8.8.3.6.1.). In the other maps with the commercial species the differences between high landings and low discards are interesting (Figure 8.8.3.6.3. to Figure 8.8.3.6.6.). However the discards of the other species will also play a role in management (Figure 8.8.3.6.2. and Figure 8.8.3.6.7.). Combinations of these maps and the using effort measurements as closing certain areas for some gears but also changes in gear specifics will result in different maps. The model can incorporate these management measures resulting in maps showing the most effective areas for the specific management measure.

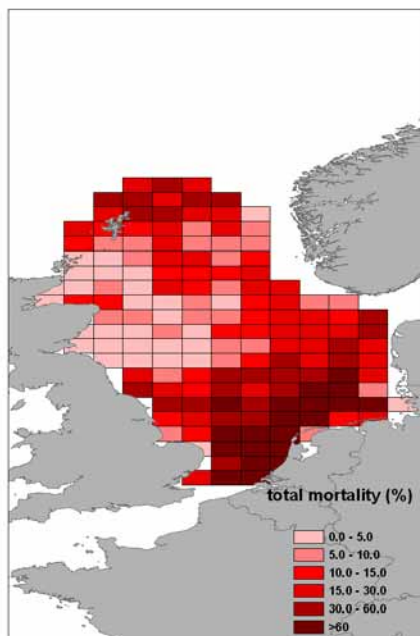


Figure 8.8.3.6.1. The total model-based mortality in percentage of the total biomass

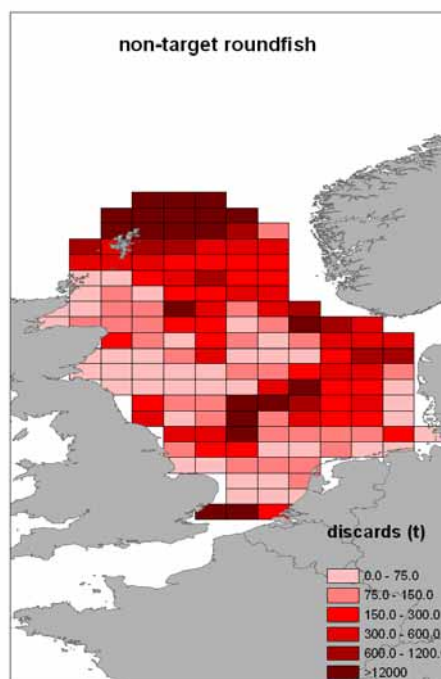
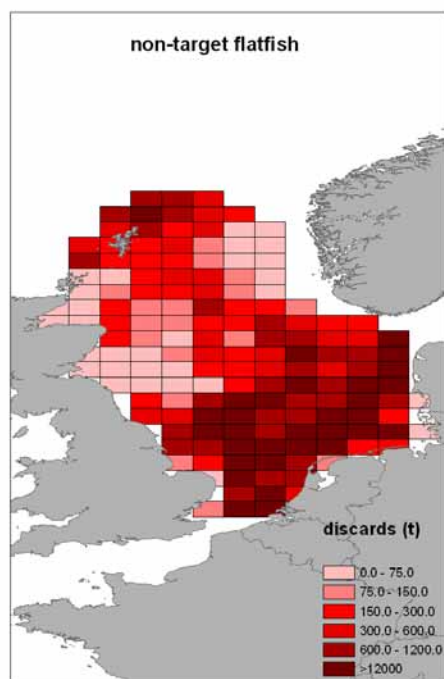


Figure 8.8.3.6.2. The model-based discards (t) of the non-target flatfish and non-target roundfish.

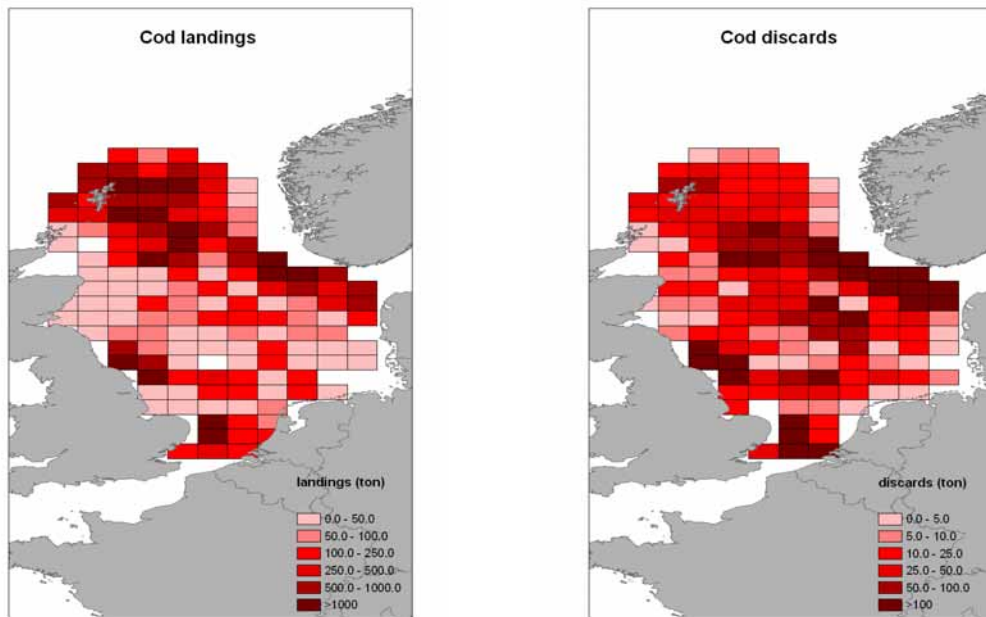


Figure 8.8.3.6.3. The model-based landings (t) and discards (t) of cod

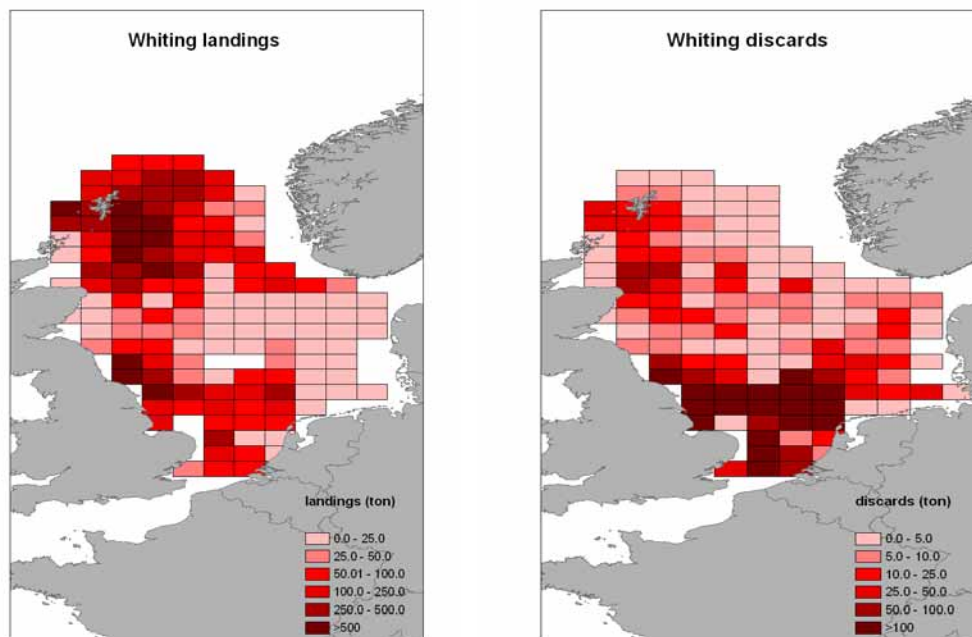


Figure 8.8.3.6.4. The model-based landings (t) and discards (t) of whiting

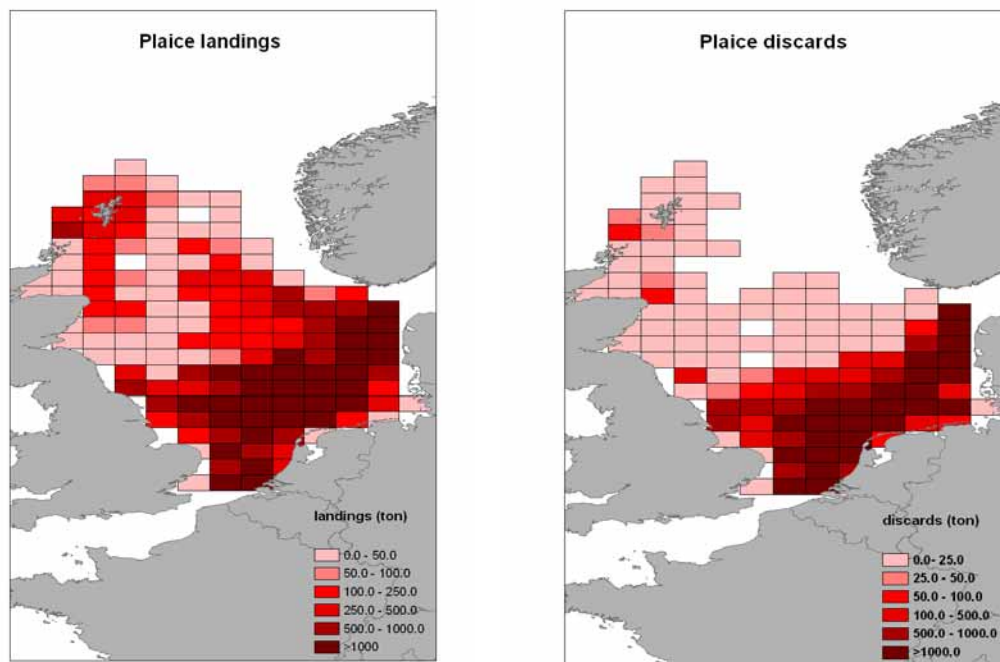


Figure 8.8.3.6.5. The model-based landings (t) and discards (t) of plaice

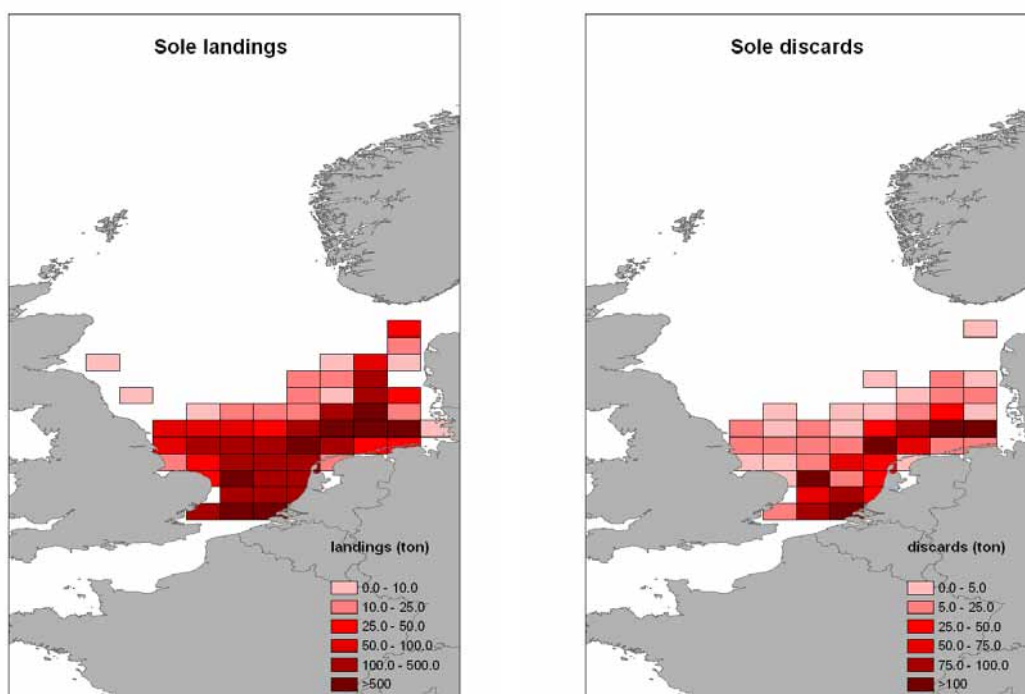


Figure 8.8.3.6.6. The model-based landings (t) and discards (t) of sole

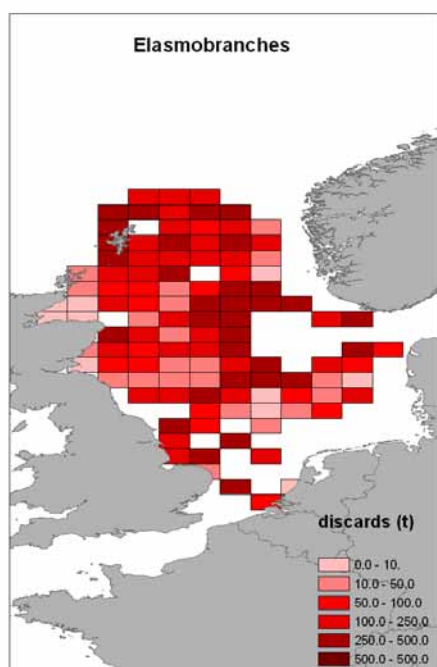


Figure 8.8.3.6.7. The model-based discards (t) of elasmobranchs

8.9. Deriving maps of ecological disturbance to fish based on landings data

The landings data provided by each of the MAFCONS institutes (described in section 8.3) took no account of discards. Discards are now estimated for all stocks as part of the assessment processes (ICES 2006), and these data enabled annual Landings to Catch raising factors to be determined for each species in each year (Table 8.9.1). These raising factors were applied to the landings data for each species in each year that were provided by the MAFCONS partners (section 8.3). Landings for the non-MAFCONS partners were modelled on CPUE derived from the MAFCONS partners' data and the catches of all species taken by each of the fleet components as each fleet component targeted particular species were estimated. As described in section 8.5, the resulting data were tantamount to actual catch data, as elements of the bycatch of species that were not specifically targeted were necessarily included. The raised (for discards) MAFCONS data were therefore simply combined with the modelled non-MAFCONS data to provide estimates of the total catch of each species in each rectangle in each year. These data were then summed over species and averaged across years, to provide estimates of the average annual biomass removal of the six demersal fish species combined from each ICES rectangle. Demersal fishing activity is not evenly distributed throughout the year; approximately 35% of demersal fishing effort occurs in quarter three. The average annual biomass removal data were therefore multiplied by 0.35 to convert them to estimates of Q3 biomass removal averaged over the eight years. Dividing these values by 92 converted them to daily removal estimates for Q3. Finally these Q3 daily removal estimates were divided by the area of each rectangle to convert them to daily catch densities for each rectangle in Q3 averaged over the period 1997 to 2004 (Figure 8.9.1). Catch densities were highest in the south-eastern and north-eastern North Sea,

Year	Cod	Haddock	Whiting	Saithe	Plaice	Sole
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1997	1.2271	1.6588	1.5501	1.0500	2.1669	1.0909
1998	1.2913	1.5843	1.5325	1.0500	2.4457	1.0909
1999	1.1307	1.6629	1.9077	1.0500	1.1783	1.0909
2000	1.2015	2.0598	1.9656	1.0500	1.3897	1.0909
2001	1.2956	4.0371	1.8494	1.0500	1.8231	1.0909
2002	1.1036	1.8472	2.1001	1.0500	1.9186	1.1667
2003	1.1977	1.5558	3.2603	1.0500	2.0125	1.1500
2004	1.1512	1.3597	2.3639	1.0500	1.9553	1.1786

Table 8.9.1. Annual raising factors to raise landings of cod, haddock, whiting, saithe, plaice and sole to catch including discards (ICES 2005).

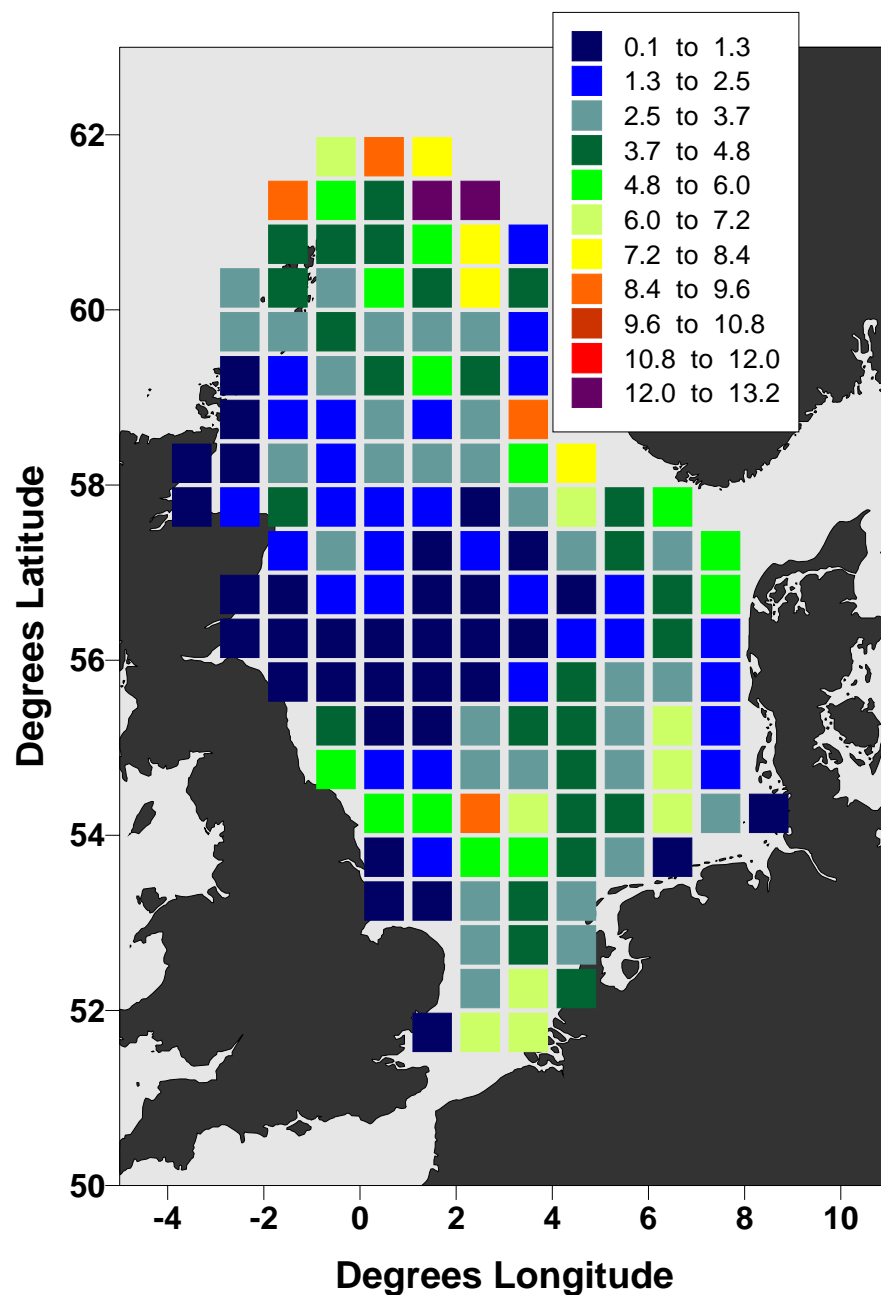


Figure 8.9.1. Daily third quarter demersal fish (cod, haddock, whiting, saithe, plaice and sole combined) catch densities ($\text{kg.km}^{-2}.\text{d}^{-1}$) averaged over the period 1997 to 2004).

In section 9.3.3.2.2 we consider the diversity of the “fished” component of the groundfish assemblage of the North Sea; fish belonging to Log_2 weight classes 8 and above. Here we wish to calculate the mortality disturbance caused by fishing to this component of the fish assemblage. Since our catch data includes discarded fish we consider that these removed fish will also include fish in Log_2 weight class 7 as well. In Section 9.3.4.2.1 we illustrate spatial variation in the biomass density of all fish assigned to various Log_2 weight classes. Figure 8.9.2 shows spatial variation in the biomass density of cod, haddock, whiting, saithe, plaice and sole combined for fish of Log_2 weight classes 7 and higher, with the three different methods of estimating density illustrated. Biomass densities were markedly higher in the northern North Sea. Dividing the daily catch densities for each rectangle in Q3 averaged over the period 1997 to 2004 by these biomass density estimates derives the required exploitation rate estimates required in Chapter 2 for testing Huston’s dynamic equilibrium model (Figure 8.9.3). Exploitation rates were markedly higher in the southern North Sea, primarily because of the differences in biomass density.

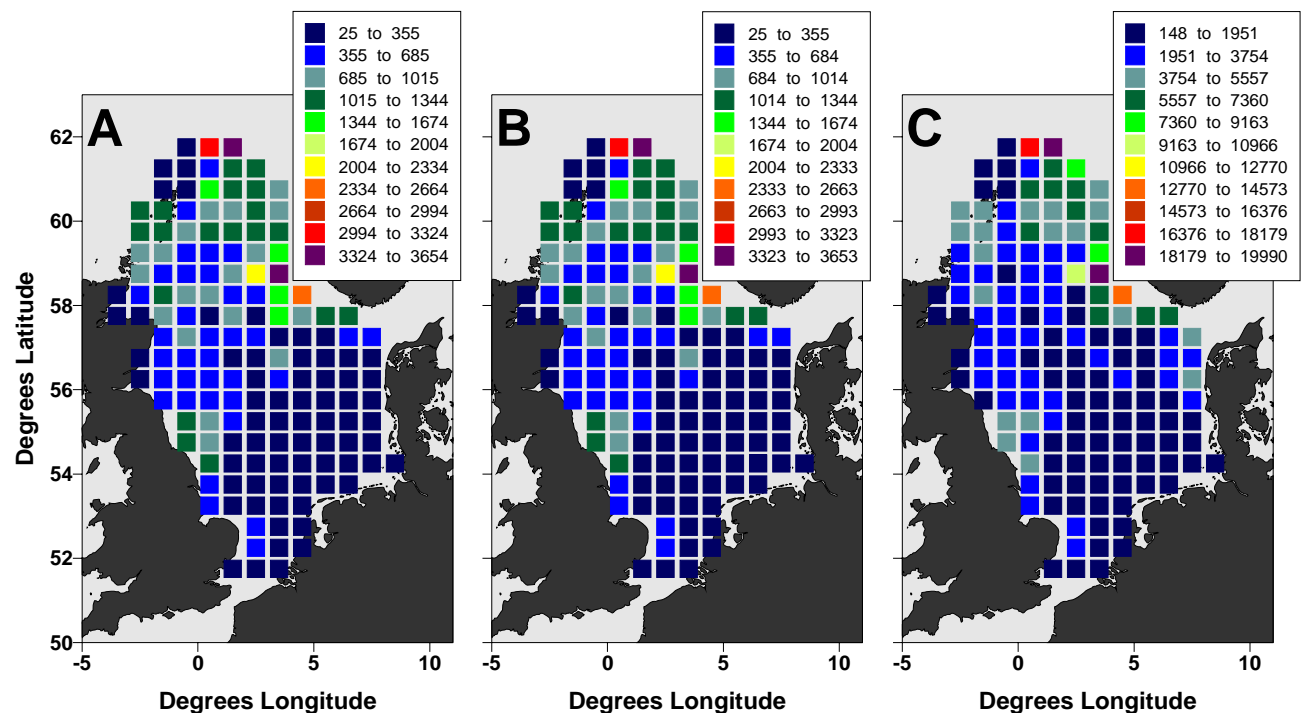


Figure 8.9.2. Spatial variation in biomass densities of cod, haddock, whiting, saithe, plaice, and sole combined belonging to Log_2 weight-classes 7 and higher averaged over the period 1997 to 2004. A: Based on arithmetic mean densities calculated across all 20 hauls in each rectangle on the raw uncorrected (for catchability) trawl densities. B: Based on geometric mean densities calculated across all 20 hauls in each rectangle on the raw uncorrected (for catchability) trawl densities. C: Based on geometric mean densities calculated across all 20 hauls in each rectangle on the raised corrected (for catchability) trawl densities.

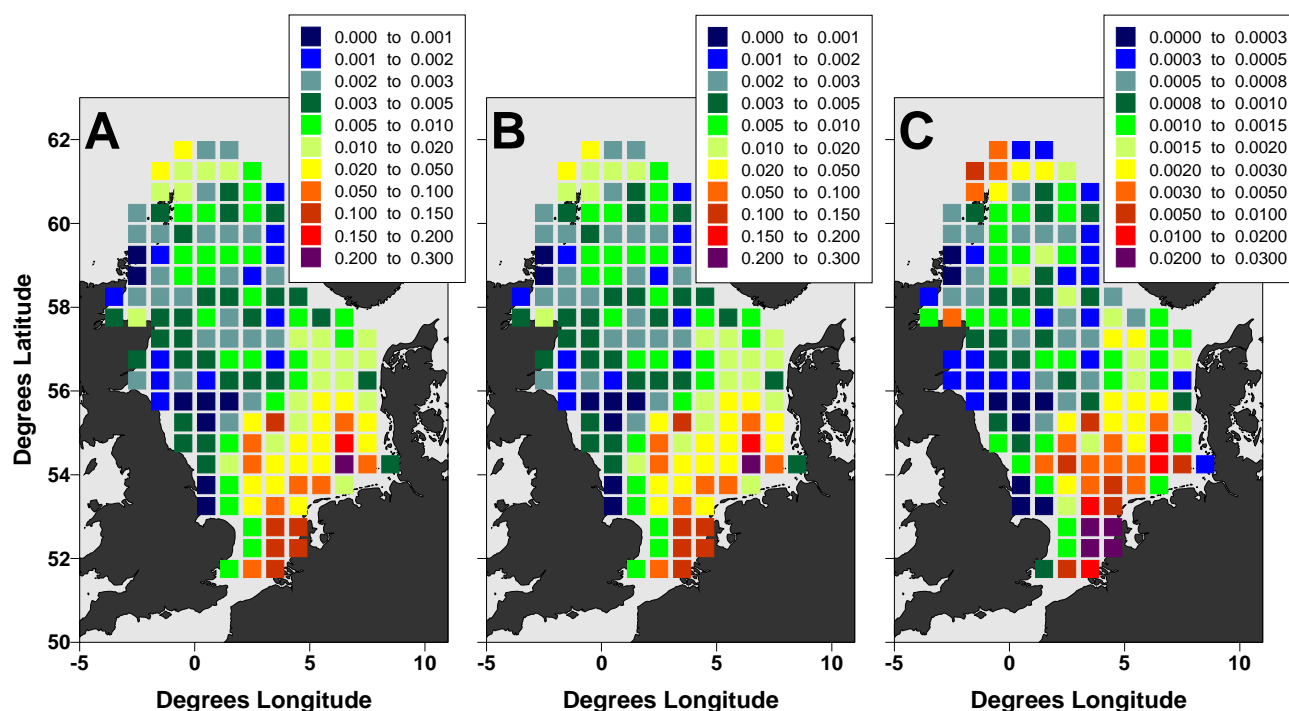


Figure 8.9.3. Spatial variation in the exploitation rate of cod, haddock, whiting, saithe, plaice, and sole combined belonging to Log_2 weight-classes 7 and higher averaged over the period 1997 to 2004. A: Based on arithmetic mean densities calculated across all 20 hauls in each rectangle on the raw uncorrected (for catchability) trawl densities. B: Based on geometric mean densities calculated across all 20 hauls in each rectangle on the raw uncorrected (for catchability) trawl densities. C: Based on geometric mean densities calculated across all 20 hauls in each rectangle on the raised corrected (for catchability) trawl densities. Note non-linear scaling.

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