# 3. The Management Protocol

## 3.1. The Original Concept

The original MAFCONS concept for managing fisheries in the North Sea so as to conserve demersal fish and benthic invertebrate species diversity was based on the premise that, through the manipulation of Total Allowable Catch (TAC) levels, the spatial distribution of fishing activity could be influenced. Huston's (1994) Dynamic Equilibrium Model suggests that the relationship between ecological disturbance and species diversity depends on local productivity. The model predicts that in regions of the North Sea where productivity is high, increased fishing disturbance would illicit an increase in species diversity. Conversely, where productivity was low, any increase in fishing disturbance would cause species diversity to decline. Therefore, the principle on which the original MAFCONS management protocol was founded was that levels of species diversity might be maintained, or even increased, by manipulating TACs to direct fishing activity towards the higher productivity regions of the North Sea, and away from regions of low productivity.

The flow chart shown in Figure 3.1.1 illustrates the original management protocol envisaged at the start of the MAFCONS project. This protocol was intended to be implemented as an additional procedure in the annual stock assessment and management advice process, rather than as an alternative to this process. Current fisheries management operates primarily through catch limitation, through the setting of TACs for each coming year. Thus the starting point for the protocol is the suite of TACs decided for each year. TACs are intended to control the mortality rates imposed on the target species by fishing exploitation. As such they provide some indication of the impending impact of fishing on the marine ecosystem. However, as indicators of the impact of fishing in the year ahead, TACs and their associated estimates of mortality are really only of relevance to the small proportion of the North Sea taxa that constitute the commercially targeted species. For by far the greater fraction of the North Sea demersal or benthic fauna, TACs provide little or no indication of the disturbance to which they will be subjected to as a result of the coming year's fishing activity. For these species, estimates of disturbance need to be derived from data that describe the perturbation itself; fishing effort statistics. Thus, determination of the patterns of fishing activity likely to occur in response to any given suite of TACs was considered to be the first step required in the original management protocol intended to predict the ecological consequences of fisheries management decisions. Relationships between landings and fishing effort therefore needed to be clearly defined and applied to the TAC input data in order to estimate the amount of fishing effort required to land these TACs, as well as its spatial distribution across the North Sea. The results of MAFCONS investigations into the relationships between landings and fishing effort are presented in Chapter 12.

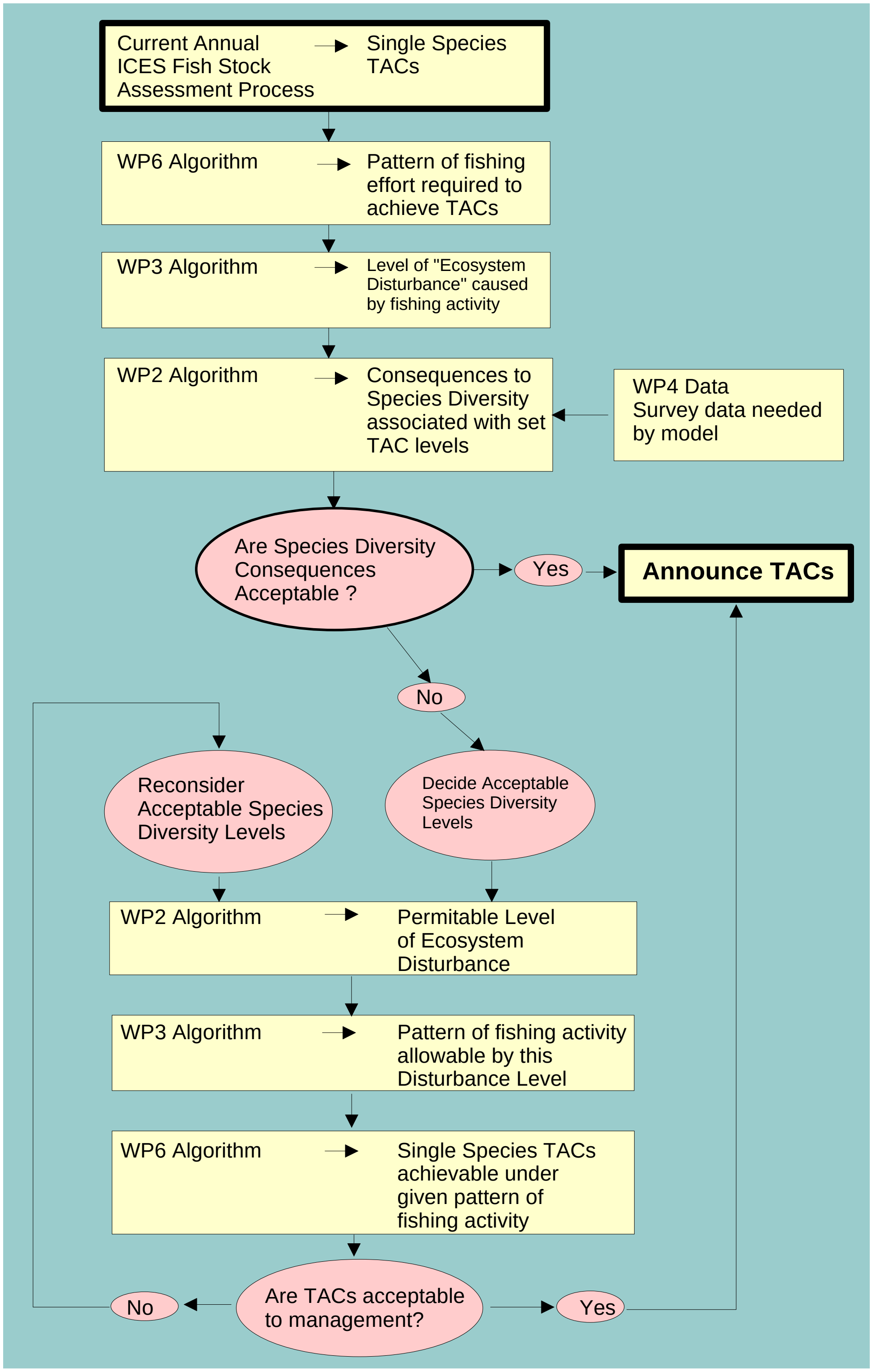


Figure 3.1.1. Flowchart for initial proposed "management protocol"

Fishing effort statistics describe the anthropogenic activity itself. In many published studies the implicit assumption is that such statistics are equivalent to the ecological disturbance caused by fishing (eg Greenstreet & Hall 1996; Jennings et al 1999; Greenstreet & Roger 2006). However, it is clear in Huston's (1994) derivation of the DEM that he considers the disturbance variable in his model to be the biological consequences of the disturbing perturbation, not the perturbation itself. In other words, it is the changes in mortality rate that occur as a result of the perturbation that alter the population dynamics of the species affected, thereby altering their competitive ability and their capacity to influence bottom and top down processes within trophically interacting species assemblages. If the relationship between fishing activity and its immediate ecological consequences (ie mortality rates), is linear (ie directly proportional) then fishing activity statistics may provide a useful proxy for ecological disturbance. However, if the relationship between these parameters is not linear, then simply using fishing effort statistics as the indicator of disturbance in the DEM could be miss-leading (eg Piet et al in Press), compromising the advice coming out of the protocol. To cover this possibility, the next step required in the original management protocol concept was the translation of spatial pattern of fishing activity to corresponding distributions of ecological disturbance. Elucidating the relationships between fishing effort and ecological disturbance for the different components of the benthic/demersal fauna of the North Sea occupied a considerable proportion of the resources of the MAFCONS consortium; both compiling the effort statistics themselves and developing the necessary biological models that relate fishing activity to the mortality caused by fishing. This work is described in detail in Chapter 8.

Having determined the patterns of ecological disturbance associated with any given set of TACs, the third step in the original management protocol concept involved inputting these data into a theoretical ecology module to predict the consequences of the proposed management action on the species diversity of the demersal fish and benthic invertebrate communities. The MAFCONS project identified Huston's (1994) DEM as the most promising candidate to form the basis of the theoretical module. The DEM has been described in detail in Chapters 1 and 2. A major part of the work undertaken by the MAFCONS consortium was directed towards the testing of hypotheses derived from the DEM to ensure that it could adequately serve this purpose. This testing of the model is reported in Chapter 2. The hope was that, should the model withstand this rigorous testing, a simple function might be parameterised that related species diversity ($S$) to both disturbance from fishing ($F$) and productivity ($P$);

$$S = f(F).f(P) \qquad 3.1.1$$

The first two steps of the management protocol would provide the input data for the disturbance part of this function. Parameterisation of the productivity aspect required the collection of new data and the development and application of new size-based methods of estimating local productivity from size-structured fish and benthic biomass data. This part of the project's work is reported in Chapters 9, 10, and 11. Finally, to fully develop the theoretical ecology module, the combined influence of productivity and disturbance on species diversity has to be determined. This involved the application of a suite of diversity indices to size-structured fish and benthos abundance data. Again this work is reported in the same three chapters.

At this point, the original concept for the management protocol has produced estimates of the species diversity consequences associated with specific fisheries management proposals (a given suite of TACs). The remainder of the protocol involves attempting to reconcile any unwelcome ecological consequences arising from the proposed fisheries regime against the economic benefits to be gained from landing the specified quantities of fish. If the estimated species diversity consequences are deemed acceptable, then there is no problem and the proposed TACs might be accepted and published. However, if this was not the case, and the species diversity consequences were considered unacceptable, then the management protocol could be re-run, either in the same direction with a new set of TACs, or in reverse with acceptable species diversity levels input and the algorithms run backwards to identify a set of TACs that was commensurate with management objectives for species diversity. TACs derived in this way may well be considered politically unacceptable in the context of supporting a viable fishing industry. Minimum acceptable TACs could then be input to determine the species diversity consequences associated with these. In effect, this original concept for a management protocol could be run and re-run in an iterative process to identify the best compromise between the needs of the fishing industry, and political obligations to conserve and restore species diversity.

## 3.2. Problems with the Original Management Protocol Concept

Over the course of the MAFCONS project it became clear that the original concept for the Management Protocol was seriously flawed. Major problems emerged in two of the steps in the protocol; the theoretical community ecology stage and the step relating TACs, landings and effort.

### 3.2.1. Problems with the DEM

In Chapter 2 we test a number of specific hypotheses derived from the DEM for explicitly defined components of the demersal fish and benthic invertebrate communities of the North Sea. In each case the model failed to explain spatial variation in species diversity. No function relating species diversity to variation in disturbance and productivity could be established in any of the analyses performed. It became obvious therefore that the DEM could not provide the theoretical basis for the community ecology stage of the Management Protocol. Following our review of the theoretical community ecology literature, presented in Chapter 7, this result was not entirely unexpected. The DEM is founded strongly in competition and niche theory. Such a model may therefore be appropriate for communities consisting primarily of species displaying deterministic growth. In such communities all mature adult individuals of any particular species have approximately the same body mass, so that species might be considered to be the fundamental ecological functional unit. In communities consisting mainly of species displaying non-deterministic growth patterns, such as demersal fish and benthic invertebrate communities in the North Sea, individuals belonging to the same species vary markedly in size. Consequently their ecological roles also differ considerably (Jennings et al 2001; 2002a; 2002b; Kerr & Dickie 2001). In such communities, individuals of similar size, regardless of species, may more appropriately be considered to be the fundamental ecological unit. Under such circumstances, the DEM is unlikely to explain variation in species diversity. In Chapter 4 we discuss these issues at much greater length.

Failure of the DEM to provide the basis for the theoretical community ecology step in the Management Protocol does not necessarily spell the end for the protocol. It simply means that the DEM is the wrong model and that an alternative is required. In Chapter 4 we develop an alternative theoretical species diversity model that is size structured and that takes account of both bottom up limitation and top down control processes and allows the impact of fishing on these processes to be modelled. Such a model, if developed, could well predict the changes in species abundance that occur as a result of varying fishing exploitation regimes, thereby allowing the effects of fishing activity on species diversity to be predicted. However, the disadvantage of the model is that it is species-specific and therefore much more demanding with respect to data requirements for parameterisation.

### 3.2.2. Problems with Catch Limitation Fisheries Management

Current fisheries management involves an annual process whereby the current state of the commercially targeted fish stocks is assessed in order to determine the level of fishing mortality that they can sustain. Once determined, these mortality rates are converted into potential catches, and these Total Allowable Catches (TACs) are divided into the individual member state quotas. Over the years, as the state of the stocks has waxed and waned, TACs have varied considerably, as the management process (including the scientific advice input) has attempted to maintain the individual fish stocks within sustainable bounds, whilst simultaneously trying minimise restrictions to fishing activity. Since 1994, there has been a clear decreasing trend in the TACs for the main demersal fish species (Figure 3.2.2.1). Prior to this, the data suggest that TAC levels were relatively stable.

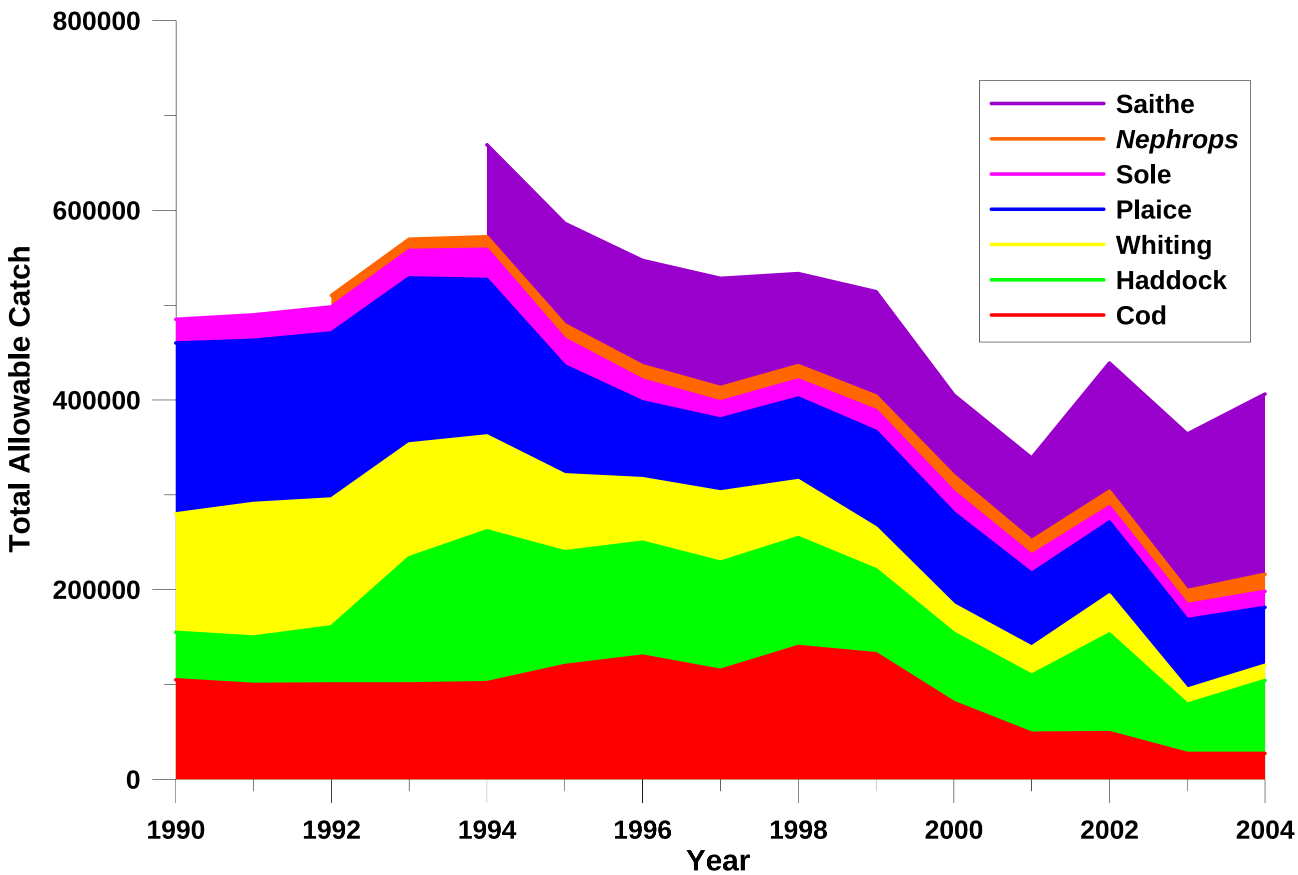


Figure 3.2.2.1. Annual variation in TAC of the main commercially targeted demersal fish stocks.

Whilst it is relatively easy to access data for North Sea landings and TACs, at least at the scale of the whole North Sea, accessing data to examine trends in fishing effort is considerably more difficult. Only two studies have attempted to compile international fishing effort databases over any appreciable period of time; the current MAFCONS study (Chapter 8) and the earlier EC funded “Biodiversity” study (Jennings et al 1999). The “Biodiversity” project covered the period 1990 to 1995, whilst the data presented in Chapter 8 cover the period 1997 to 2004. When data for the two periods were plotted together the data suggest that levels of fishing effort only started to decline from around 1997 onwards (Figure 3.2.2.2A). However, these two studies did not include participants from all nations operating fisheries in the North Sea. In Chapter 8 we have attempted to model fishing effort data for the four countries with significant North Sea quotas that did not contribute to the MAFCONS effort database, Denmark, France, Sweden, and Belgium. Trends in total fishing effort suggest that effort really only started to decline in the North Sea from around 2000 onwards with the introduction of decommissioning and limitation of the numbers of days absence from port each month (Figure 3.2.2.2B). These conclusions seem a little tenuous because of the short duration of the data available. However, Scottish effort data covering the period 1960 to 2004 have been analysed and these certainly indicate a sharp decline in fishing effort since 1998 (Greenstreet et al 1999; 2006). The increase in otter trawl through to the late 1980s was mainly associated with decreasing use of seine gear, with fishing vessels switching to otter trawl so that, by the late 1990s, otter trawl was the principal gear used by Scottish fishermen. The implication from this assessment of the available fishing effort data is that reductions in TAC had little or no effect on fishing effort levels until moves to control fishing effort directly were introduced. At the very best, a lag of several years was apparent following reduction in TACs before fishing effort levels started to decline. In Chapter 12 we explore the relationships between TACs, landings and fishing effort explicitly. We come to the conclusion that it would be difficult to estimate the spatial distribution of fishing effort associated with any given set of TACs with sufficient precision as to provide adequate input into the next stage of the Management Protocol. Consequently, the level of uncertainty involved with any advice regarding the species diversity consequences associated with each set of TACs would be too high to be of value for management purposes.

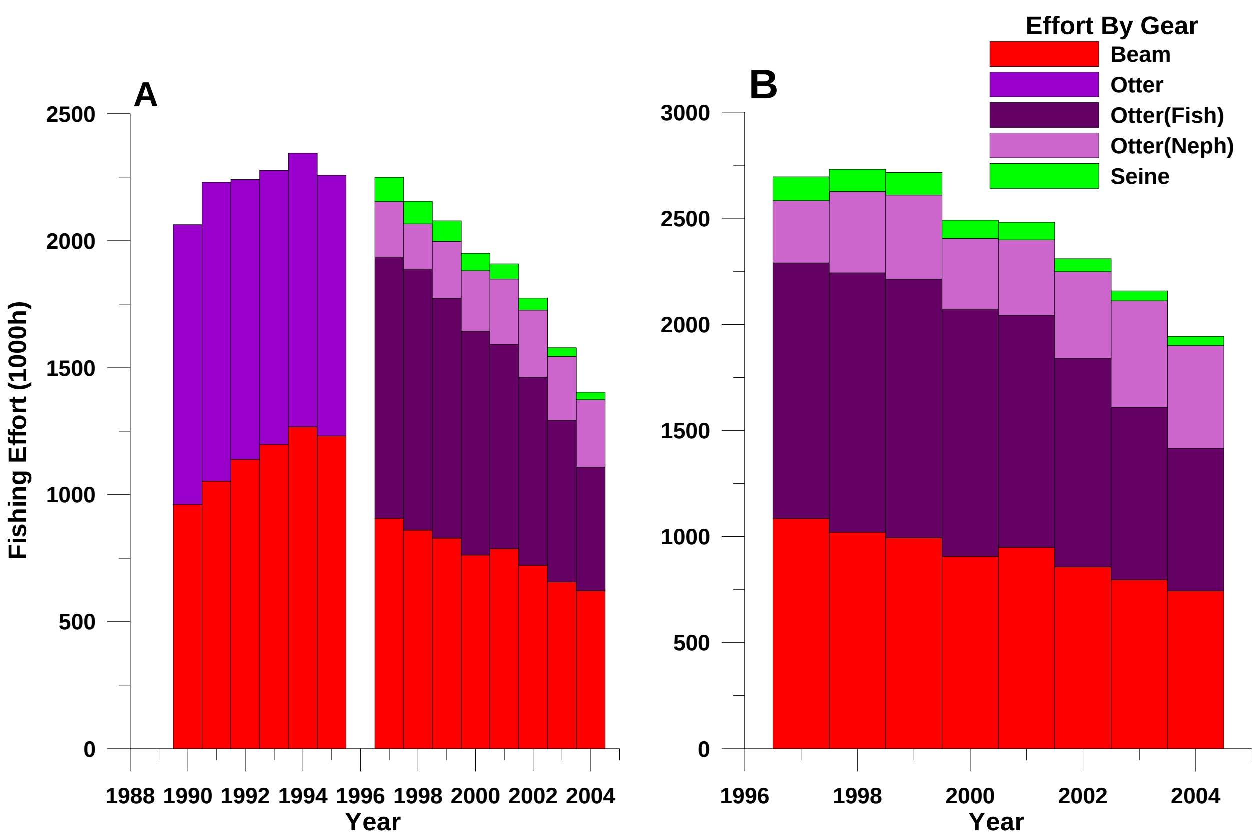


Figure 3.2.2.2. Temporal trends in international fishing effort in the North Sea. A: Data submitted to the Biodiversity project (1990 to 1995) and to the MAFCONS project (1997 to 2004) showing only the data submitted by the participating countries (1990 to 1995 excludes France, Belgium and Sweden; 1997 to 2004 excludes Denmark, France, Belgium and Sweden). B: Total North Sea effort between 1997 and 2004 including modelled estimates of effort by the four countries that did not contribute to the data base.

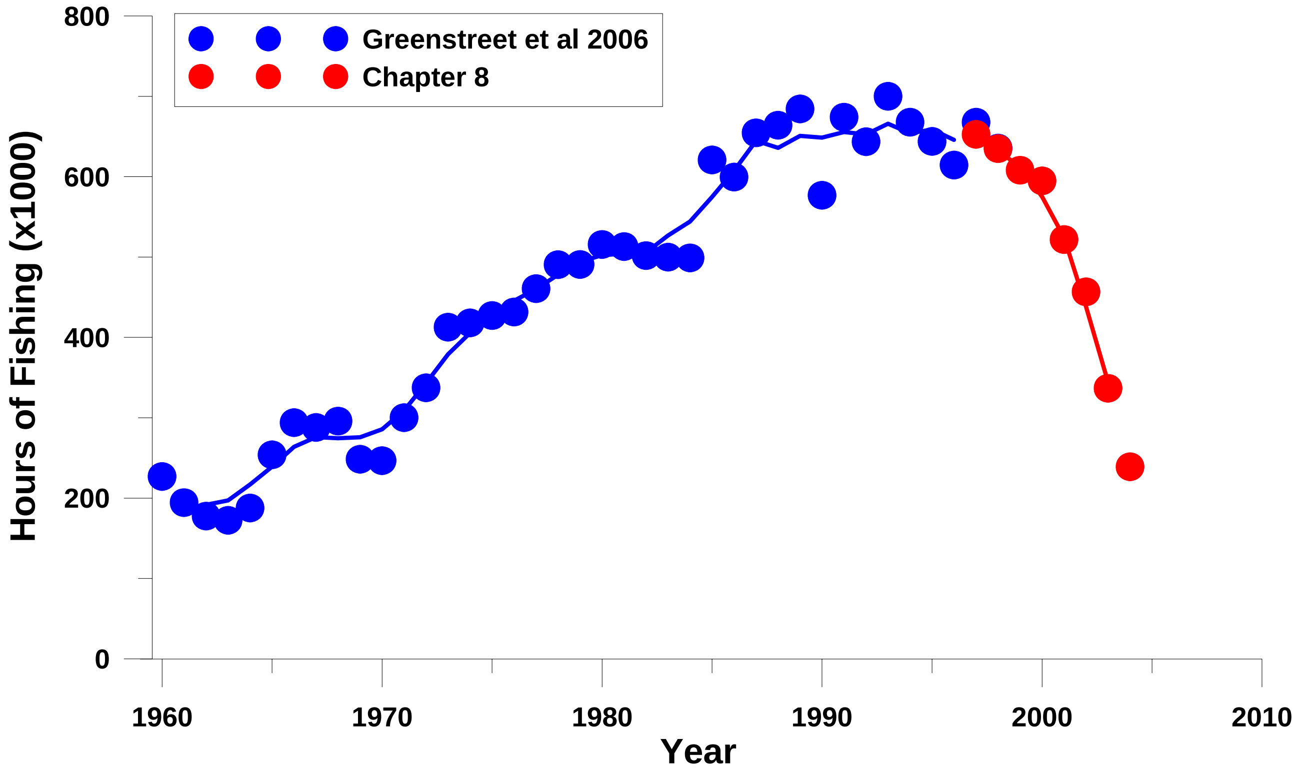


Figure 3.2.2.3. Long-term trends in Scottish Otter Trawl Effort derived from two separate studies. Lines show moving average fits to the two data sets.

A further consideration was the fact that whilst TACs may serve to limit officially recorded landings, their relationship to actual catches was far more obscure. For years now official landings data have failed to represent the actual quantities of fish landed because of the illegal landings of fish caught over-quota. In addition, the wide-spread practice of discarding under-sized fish, fish of lower value, over-quota fish and fish of no commercial value further extends the difference between the actual amount of fish caught, and hence the effort required to catch them, and the original TACs set by management. Discarding is an inevitable consequence of TAC/quota based management, especially in situations involving mixed-species fisheries, the situation prevalent throughout the North Sea.

Finally, TACs and fishing effort levels tend to be decoupled because of confounding variation in stock size. TACs tend to decrease when stock sizes decline so as to control rates of fishing mortality. But as stocks become less abundant, so they become more difficult to catch, and catch per unit effort (CPUE) declines. Consequently, the most common situation would be for TACs and CPUE to decrease in tandem, thus requiring the same, or even increasing, levels of fishing effort to take smaller and smaller catches.

## 3.3. An Alternative Approach for Management

A major assumption underpinning the original concept for a Management Protocol was that in order to achieve ecological objectives as part of an ecosystem approach to fisheries management, not only would the amount of fishing activity need to be controlled, but also its spatial distribution. The fundamental principle at the heart of the protocol was that fishing activity should be directed away from areas considered to be the most vulnerable or most valuable in terms of their intrinsic ecological worth, and shifted where necessary to less vulnerable areas or to areas of less value. Rather than attempting to do this through the manipulation of TACs, this can be achieved directly through the establishment of Marine Protected Areas (MPAs) (Pikitch et al 2004; Norse et al 2005).

Globally Marine Protected Areas (MPAs) are a commonly used marine management tool to protect commercial fish stocks, particular marine species, communities and habitats (Carr & Reed 1993; Allison et al 1998; Houde 2001; Roberts et al 2001; Botsford et al 2003; Gerber et al 2003). In recent years within Europe, increased awareness of the importance of the environment in maintaining the general health of marine ecosystems has generated considerable interest in the establishment of MPAs. The use of Marine Protected Areas (MPAs), in this context taken to mean areas closed to fishing, to achieve management objectives for broader marine ecosystem issues has been explicitly proposed in policy drivers such as the Convention on Biological Diversity, the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR), The World Summit on Sustainable Development, The 2002 Bergen North Sea Ministerial Conference, the EC Habitats, Birds, and Marine Strategy Directives, and UK Marine Strategy documents such as The Review of Marine Nature Conservation, Net Benefits (The PM's strategy unit 2004), "Charting Progress" and "Seas the Opportunity". Application of the Habitats Directives to the offshore zone will lead to the establishment of new MPAs around European coasts designed to protect vulnerable seabed habitats (EEC 1992). Under OSPAR, nations are further obliged to establish a coherent network of MPAs by 2010. The forthcoming UK Marine Bill, for example, is likely to feature a new mechanism to enable further MPAs to be established to protect features of national importance.

As pressure to designate MPAs increases, several important issues need examination. Firstly, how successful are MPAs in delivering their ecological objectives? Currently it is not clear precisely what closed areas can and can not deliver. World-wide there is certainly a considerable body of evidence to suggest that MPAs are beneficial at the local scale, ie within the boundaries of the designated closed area (Halpern 2003), but data indicating that MPA are useful in achieving global scale (eg North Sea wide) management objectives are scarce. Since the majority of MPA sites will be subject to some form of fisheries restrictions, one major concern focuses on the effects of fishing activity displaced from areas closed to fishing to alternative locations. There is a real risk that in some situations such displaced fishing activity may have serious unintended consequences, perhaps even resulting in net losses for the marine ecosystem rather than gains. This raises the possibility that particular closed area proposals may not achieve the ecological gains anticipated, thereby proving inadequate to meet specified global scale management objectives.

A greater understanding of the circumstances whereby the establishment of MPAs results in net ecological benefits, and the scale of these benefits, is required in order to support evidence based policy making and management. Scientists need the tools to allow them to assess the costs and benefits, both ecological and economic, of each individual MPA proposal, in order to evaluate the sort of large scale ecological objectives that might best be addressed through closed area management. Parts of the original Management Protocol concept can be developed to meet this requirement, particularly step two, the fishing activity and disturbance modeling stage. The wealth of information obtained during the MAFCONS project regarding spatial variation in the abundance, biomass, production and species diversity of the fish and benthic infauna across the North Sea will be invaluable in identifying potential MPA locations. The comprehensive total international effort and landings data sets compiled during the MAFCONS project can be interrogated to assess the consequential costs of closing particular areas to fishing activity. For a specified area, the loss of access to potential landings of each species by each of the countries operating in the North Sea can be estimated. If TACs are not reduced by equivalent amounts, it is anticipated that these landings will be made up by fishermen relocating to alternative grounds. If these new grounds are less productive, this may cause overall effort to increase. Knowing landings per unit effort at each ICES rectangle, this increase in effort can be quantified. It is entirely possible that any increase in overall effort, particularly when displaced to areas that previously might have only been lightly fished, may actually cause ecological damage that outweighs any gains achieved from the MPA. The ecological disturbance models can be used to determine the ecological damage associated with this displaced fishing activity, so enabling ecological costs-benefits analysis to be undertaken.

### 3.3.1. Protocols to Assess the Potential of MPA Proposals

Set aside for nature conservation purposes in terrestrial systems amounts to approximately 8 to 10% of total land cover. If MPAs were to cover a similar percentage of the North Sea, then this would equate to closing some 17 to 22 ICES statistical rectangles to fishing. For the purposes of our demonstration, we therefore assume that up to 20 ICES rectangles will be closed to fishing in order to achieve specific ecological objectives. We now demonstrate how the databases and disturbance models developed during the MAFCONS project might be used to ascertain the most appropriate rectangles for closure and to explore the consequences of the closures on various aspects of the marine ecosystem of the North Sea.

As already discussed, a major concern over the designation of MPAs is what happens to the fishing activity that would normally have occurred in the closed area? If displaced to alternative areas, then where to? If these areas are less productive, does this result in increased effort. From a management perspective it is essential to know that the benefits derived from the establishment of each MPA are not outweighed by the detrimental consequences arising from this displaced fishing activity. If the ecological costs associated with the establishment of a particular MPA outweigh the anticipated benefits, then this may seriously call into question the wisdom of using closed area management to address the issue concerned. Alternatively action might also be taken to mitigate against the disadvantages by introducing steps to reduce fishing activity displacement. Such measures might, for example, include concomitant reductions in TACs associated with specific MPA proposals. In which case it would be important to know which TACs should be reduced, and by how much? Again the data sets and models developed during the MAFCONS project can be used to address questions such as these.

For the purposes of our demonstration of this alternative Management Protocol, we consider two potential ecological quality objectives that managers might well be required to address in the near future. These are:

- Use MPAs to protect those regions of the North Sea where the species diversity of the demersal fish community is highest.
- Use MPAs as a means of reducing the overall impact of fishing on the benthic invertebrate community by 30%.

#### 3.3.1.1. *MPAs and the Conservation of Fish Species Diversity*

For this case study our objective is to select up to 10% of the area of the North Sea that might be considered to have the highest conservation value in terms of its fish community species diversity and to consider the implications of setting this area aside as MPAs with the complete abolition of fishing within them. Firstly we only have species diversity data for 152 ICES statistical rectangles covering a total area of 496,424Km$^2$, thus the total extent of our closed areas should amount to around 15 ICES rectangles covering an area of approximately 49,000Km$^2$. The data we used to guide our selection of closed areas is the GOV Q3 IBTS data set raised to account for catchability. In Chapter 9, Figures 9.3.3.2.1.3 and 9.3.3.2.1.4 suggest that the fish communities occupying the northern and southern halves of the North Sea are quite distinct, with each community differing markedly in species diversity. Under these circumstances we would wish to protect "examples" of both types of fish assemblage, and so would not locate all the MPAs in the most species diverse region, the southern North Sea. Figure 3.3.1.1.1 shows the spatial demarcation of the main assemblage types across the North Sea along with a plot of spatial variation in Hills $N_1$. In selecting ICES rectangles to set aside as MPAs to protect species diversity we did not simply select the 7 or 8 most species diverse rectangles in each assemblage type, instead we attempted to select groups of rectangles with relatively high species diversity that were contiguous with one another. Our intention here was to reduce the number of individual MPAs, and increase the size of each area. We believe this to be a better design strategy for the conservation of species diversity (Halpern 2003; Neigel 2003), more easily enforced, and perhaps of greater benefit in terms of the export of fish out of the MPAs into areas where they might be exploited. Figure 3.3.1.1.1 shows the 14 rectangles selected as MPAs, covering a total combined area of 46,595Km$^2$, or 9.4% of the area surveyed by the GOV.

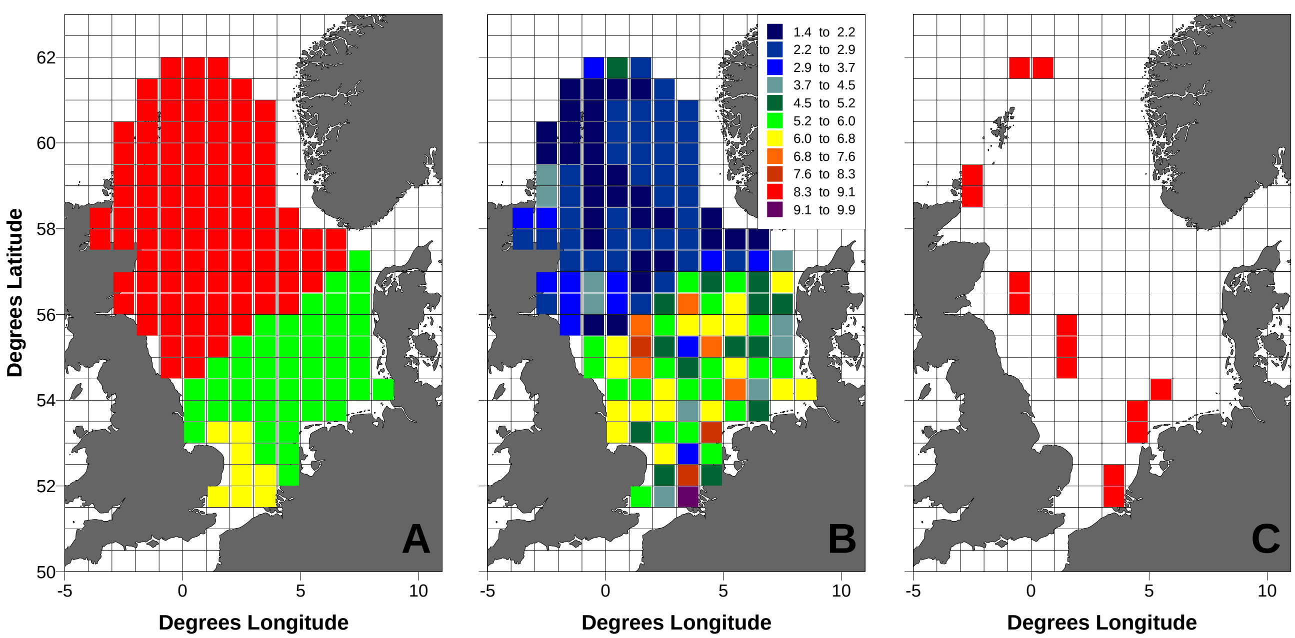


Figure 3.3.1.1.1. Spatial variation in groundfish species assemblage type (A) and Hills $N_1$ based on the GOV data raised to account for catchability (B) and the selection of approximately 10% of the surveyed area as MPAs designed to conserve the most species diverse regions within the area occupied by the two main assemblage types (C).

Landings and effort data were determined for 215 ICES statistical rectangles in ICES area IV (the North Sea), covering an area of 608,122Km$^2$ (Chapter 8). The 14 rectangles selected as MPAs to conserve groundfish species diversity constitute 7.7% of this total area. Only small quantities of

Nephrops were landed from these MPA rectangles. The proportion of total landings of cod, haddock and whiting originating from the proposed MPA rectangles, at 3.7%, 3.5% and 4.2% respectively, were considerably less than the expected 7.7%, but the proportions of saithe, plaice and sole, at 11.7%, 10.9% and 23.0% were markedly higher (Figure 3.3.1.1.2A) . Not surprisingly, given these figures, the beam trawl fishery would be the most affected by the closure of these 14 rectangles to fishing; 22% of the total annual beam trawl effort occurred in the areas selected as possible MPAs. Other métiers were much less affected with only 4.1% of otter trawl effort directed at fish, 0.2% of otter trawl effort directed at *Nephrops*, and 5.5% of seine gear effort on average occurring in the 14 selected rectangles (Figure 3.3.1.1.2B).

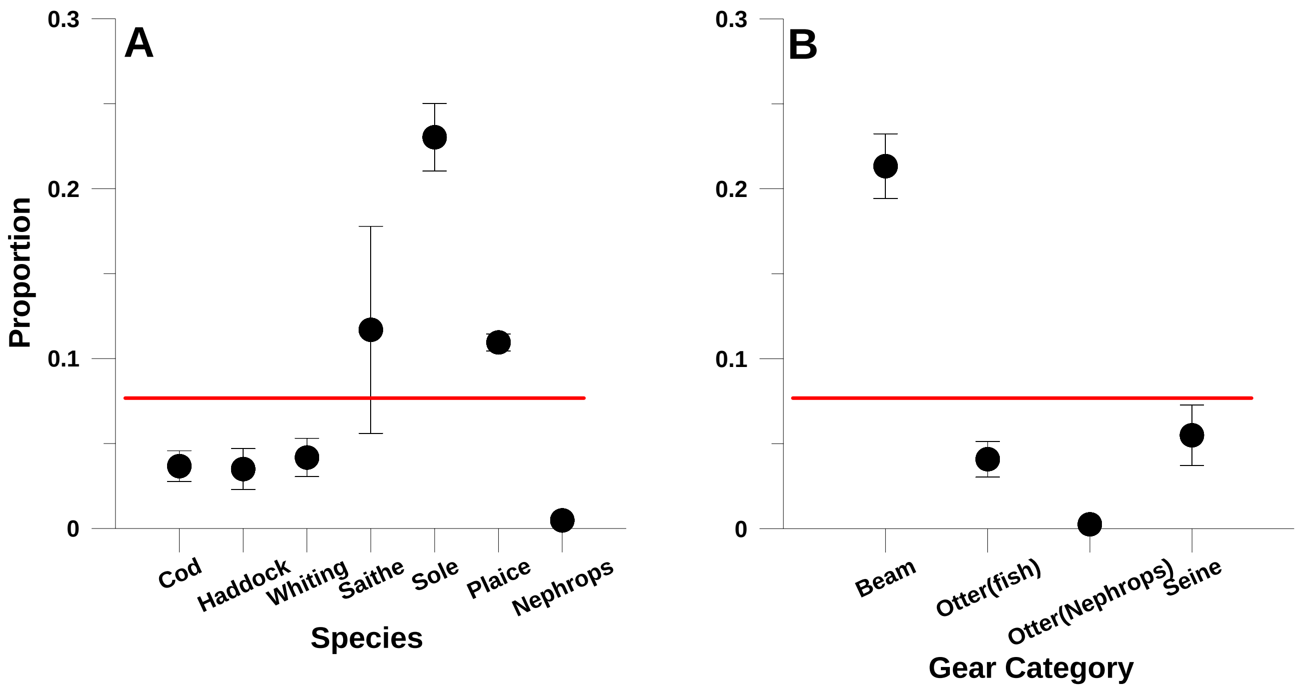


Figure 3.3.1.1.2. Mean proportion (±1 Std.Dev.) of total North Sea landings of each species (A) and fishing effort for each main gear category (B) reported from 14 ICES rectangles selected as MPAs to conserve groundfish species diversity over the period 1997 to 2004. Red lines indicate the expected fractions given the fraction of the area of the North Sea included within the proposed MPAs.

To avoid the displacement of this fishing effort to areas outside the proposed MPA would require a simultaneous reduction in TAC by the percentages indicated in Figure 3.3.1.1.2A). Without this reduction in TACs, then fishing effort is likely to be displaced to ICES rectangles remaining open to fishing outside those designated as MPAs, so as to make up the shortfalls in landed fish normally taken from within the MPA designated rectangles. The spatially referenced international landings and effort data compiled by the MAFCONS project allow this displaced effort to be modelled. To demonstrate this model we consider two time periods, 1997 to 2000 and 2001 to 2004, and use averaged annual data for these two periods. Spatial distributions of average annual landings of each species by each main gear category for these two periods are given in Chapter 8, Figures 8.6.1.2.1 to 8.6.1.2.7. Spatial distributions of effort by each main gear category in each year are also given in Chapter 8, Figures 8.6.2.2.1 to 8.6.2.2.4. From these latter data, spatial distributions of averaged annual spatial variation in effort for each of the two time periods are easily calculated. From these two sets of data, average annual catch per unit effort (CPUE) for each species in each main gear category can be determined (Figures 3.3.1.1.3 to Figures 3.3.1.1.9).

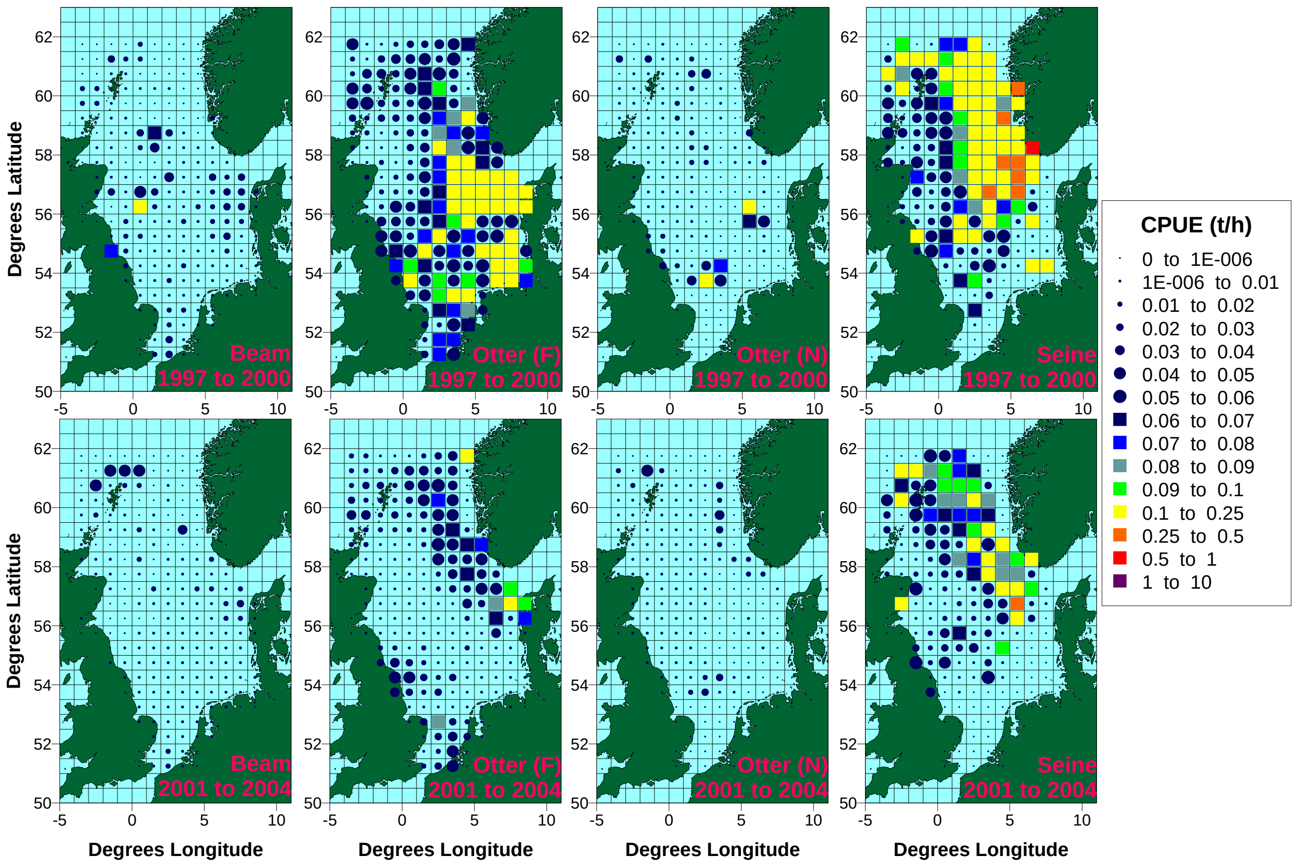


Figure 3.3.1.1.3. Spatial variation in average annual CPUE ($t.h^{-1}$) of cod in each of the main fishing gear categories (F indicates otter trawl directed at fish, N indicates otter trawl directed at *Nephrops*) in two time periods, 1997 to 2000 and 2001 to 2004.

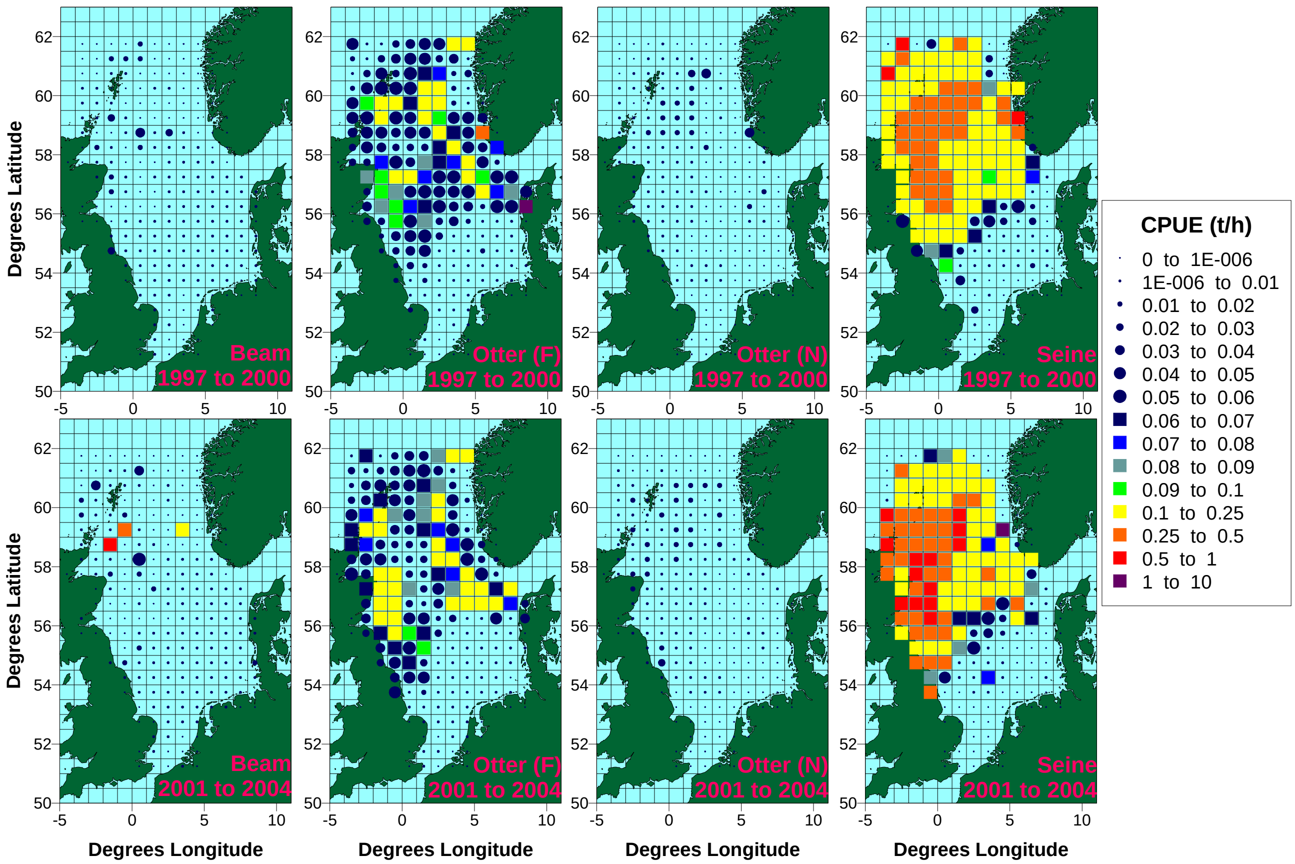


Figure 3.3.1.1.4. Spatial variation in average annual CPUE ($t.h^{-1}$) of haddock in each of the main fishing gear categories (F indicates otter trawl directed at fish, N indicates otter trawl directed at *Nephrops*) in two time periods, 1997 to 2000 and 2001 to 2004.

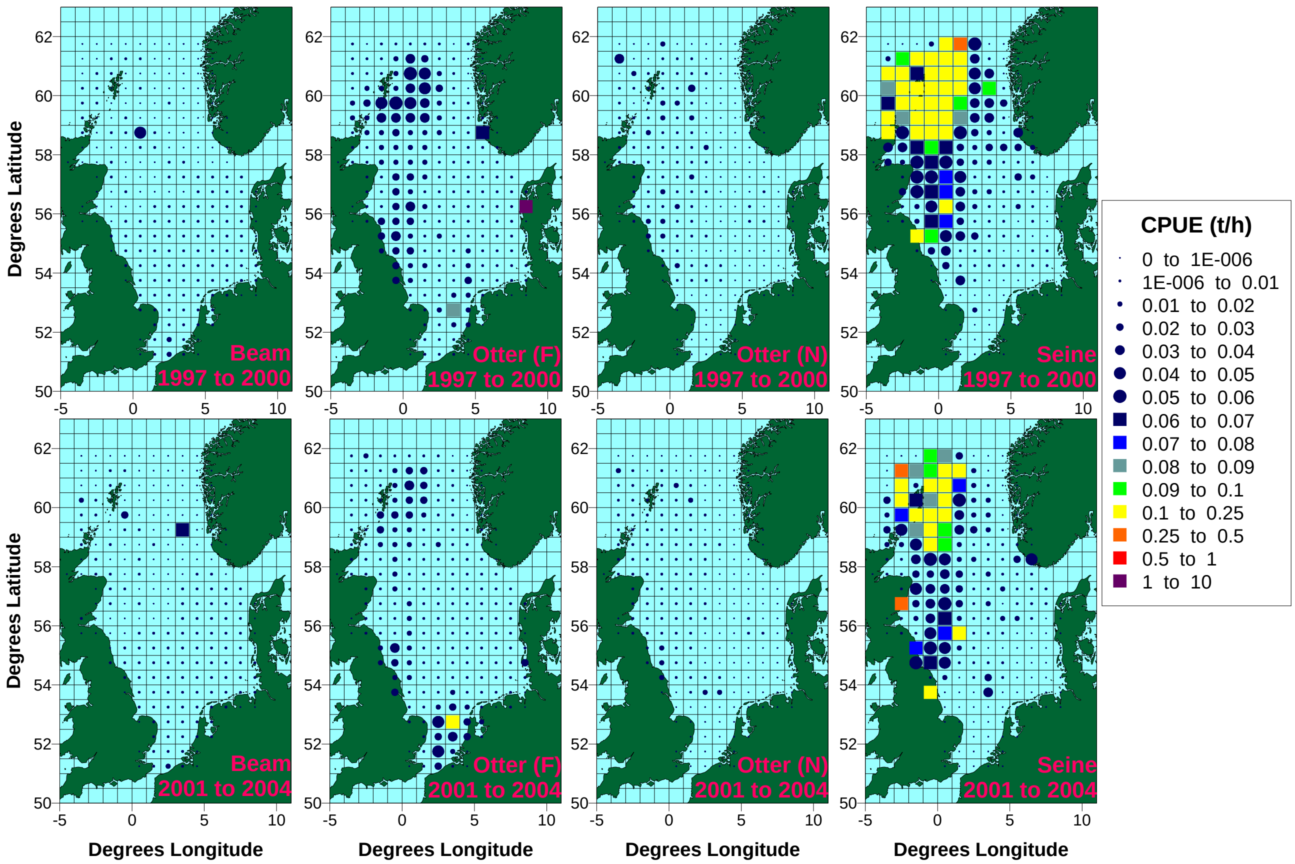


Figure 3.3.1.1.5. Spatial variation in average annual CPUE ($t.h^{-1}$) of whiting in each of the main fishing gear categories (F indicates otter trawl directed at fish, N indicates otter trawl directed at *Nephrops*) in two time periods, 1997 to 2000 and 2001 to 2004.

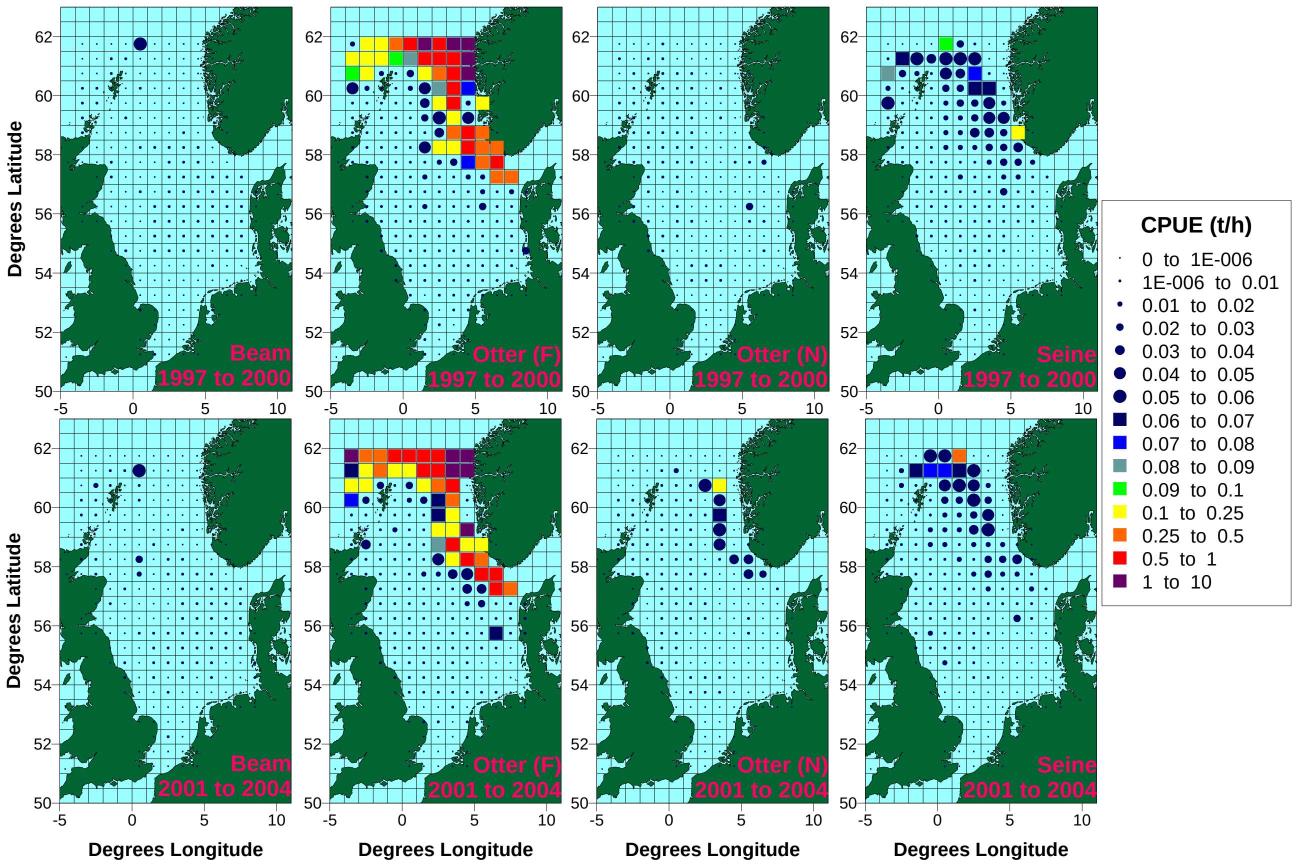


Figure 3.3.1.1.6. Spatial variation in average annual CPUE (t.h$^{-1}$) of saithe in each of the main fishing gear categories (F indicates otter trawl directed at fish, N indicates otter trawl directed at *Nephrops*) in two time periods, 1997 to 2000 and 2001 to 2004.

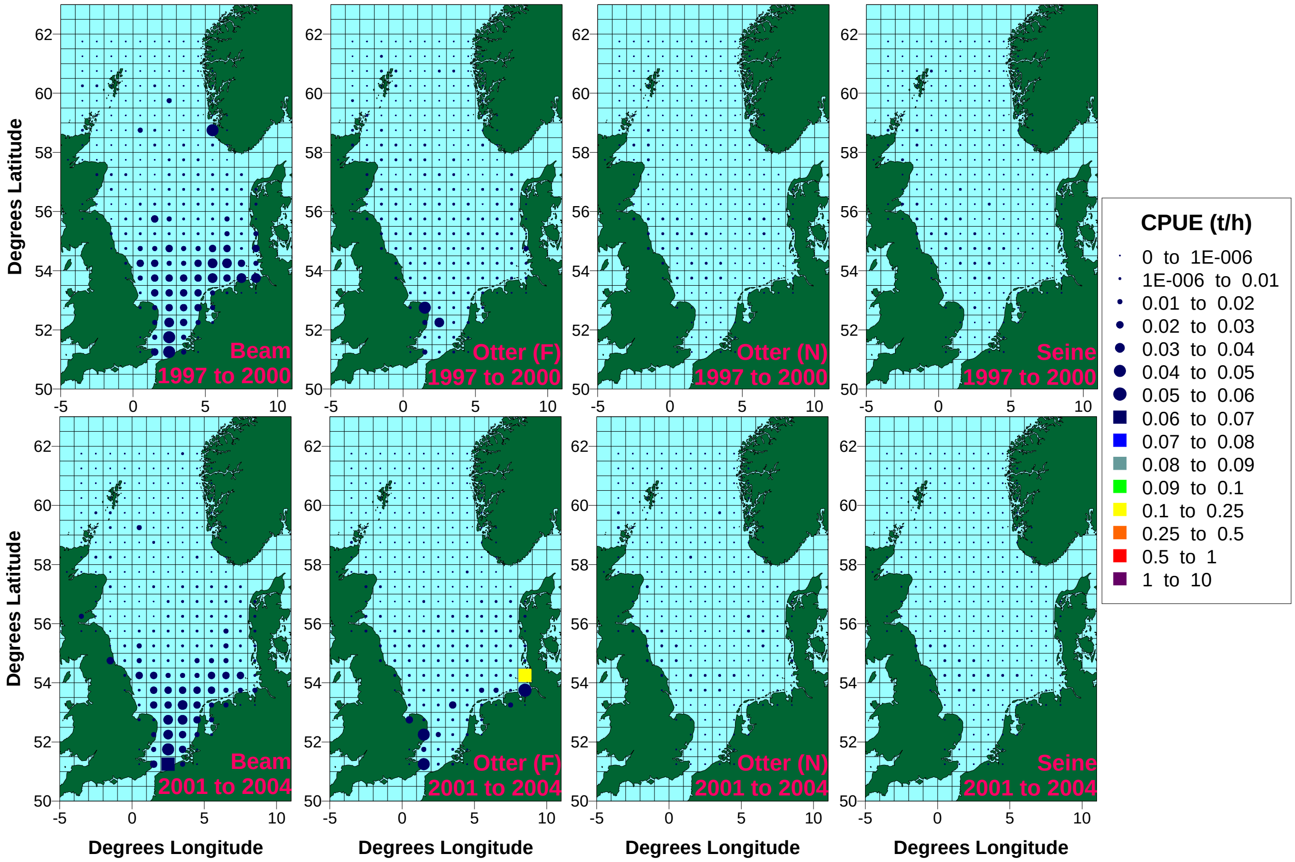


Figure 3.3.1.1.7. Spatial variation in average annual CPUE ($t.h^{-1}$) of sole in each of the main fishing gear categories (F indicates otter trawl directed at fish, N indicates otter trawl directed at *Nephrops*) in two time periods, 1997 to 2000 and 2001 to 2004.

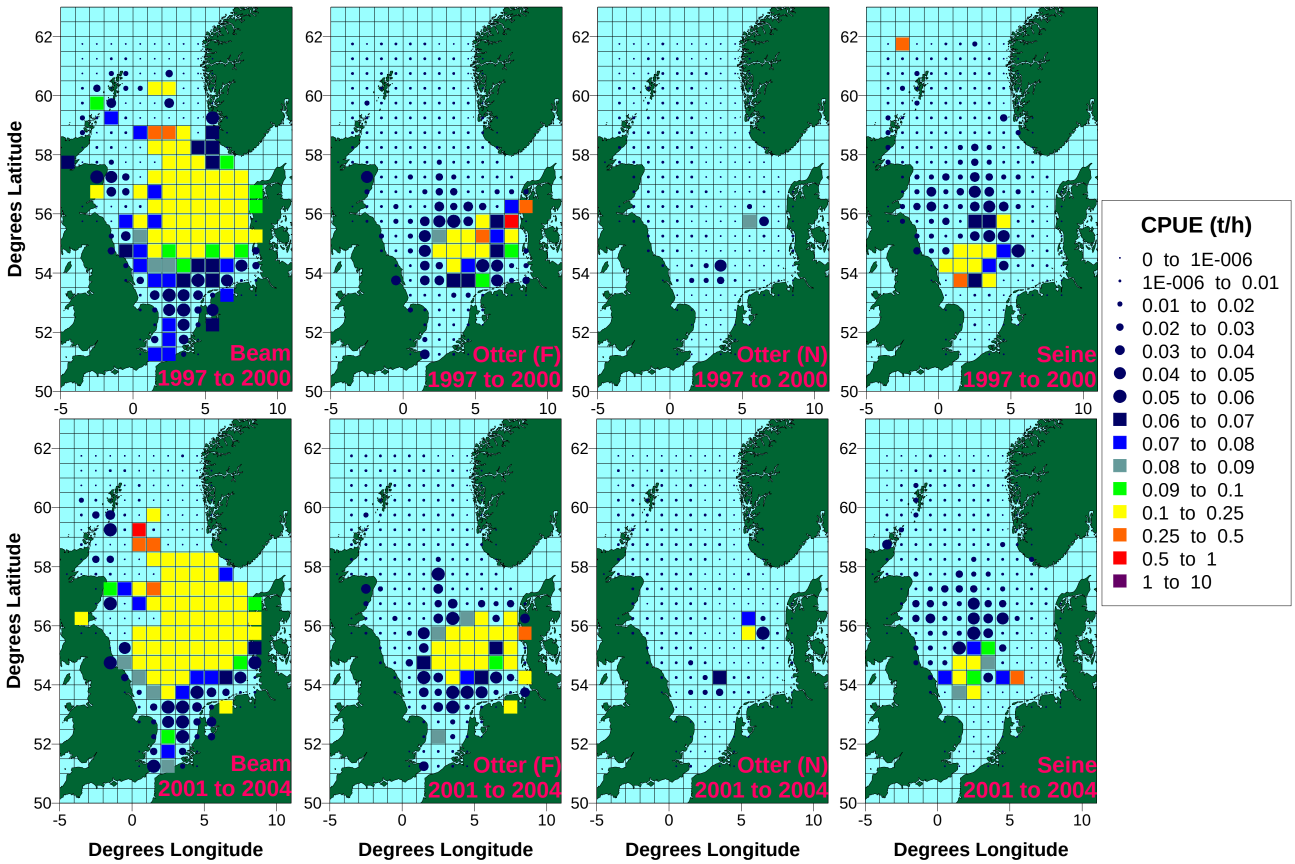


Figure 3.3.1.1.8. Spatial variation in average annual CPUE ($t.h^{-1}$) of plaice in each of the main fishing gear categories (F indicates otter trawl directed at fish, N indicates otter trawl directed at *Nephrops*) in two time periods, 1997 to 2000 and 2001 to 2004.

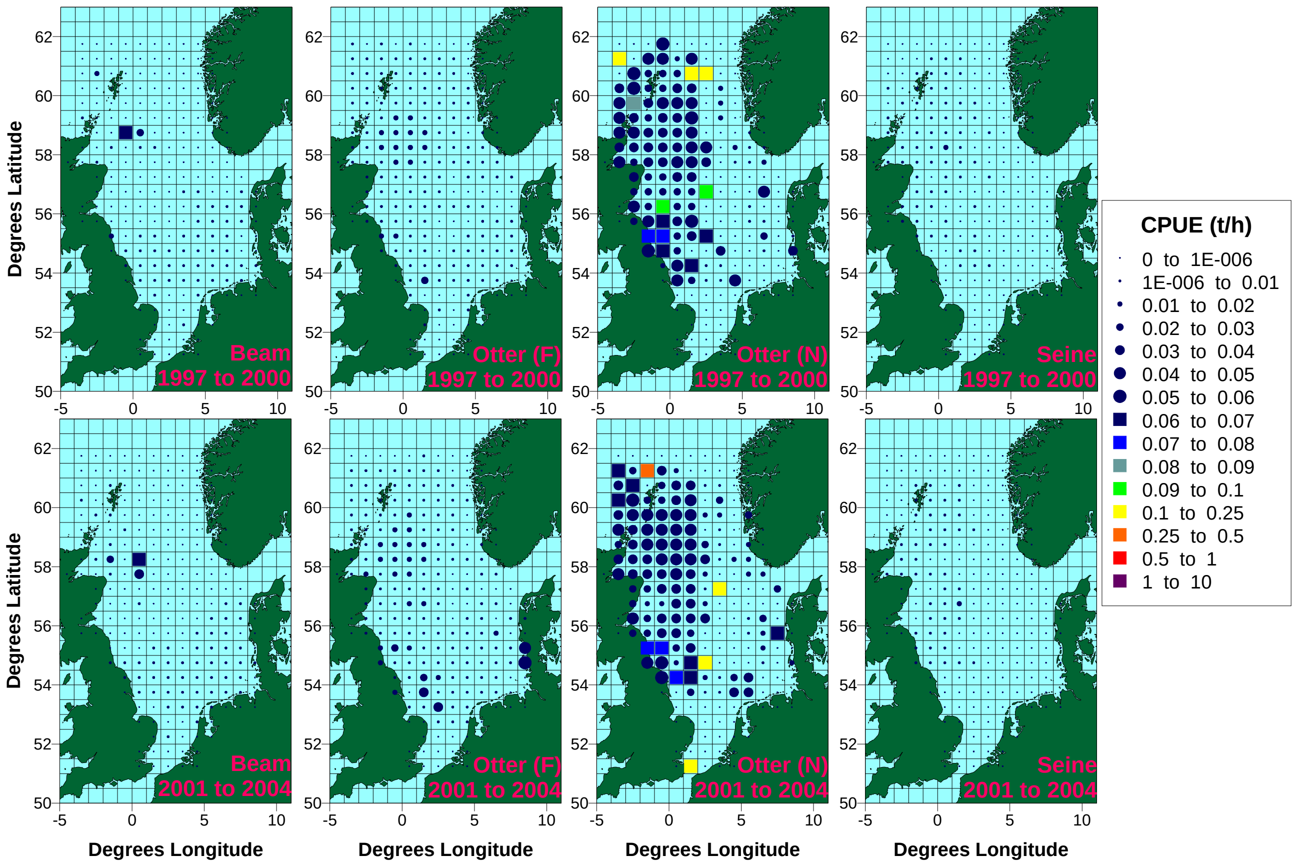


Figure 3.3.1.1.9. Spatial variation in average annual CPUE ($t.h^{-1}$) of *Neprops* in each of the main fishing gear categories (F indicates otter trawl directed at fish, N indicates otter trawl directed at *Nephrops*) in two time periods, 1997 to 2000 and 2001 to 2004.

The first step in modeling the redistribution of fishing effort, following closure of ICES rectangles designated as MPAs, is to estimate the additional landings of the principal species (*sg*) targeted by each main gear category (*g*) from each rectangle (*r*) remaining open to fishing required to balance the total quantity of that species normally landed from the closed rectangles prior to their closure. The model assumes that the total amount of landings normally originating from the MPA rectangles is, following their closure, now obtained from the rectangles remaining open to fishing *pro rata* to landings from these rectangles prior to the establishment of the MPAs, so that:

$$L_{sg,g,r,add} = \frac{L_{sg,g,r,prior}}{\sum_{r=x}^{OPEN} L_{sg,g,r,prior}} . \sum_{r=y}^{MPA} L_{sg,g,r,prior} \qquad 3.3.1.1.1$$

$L_{sg,g,r,add}$ and $L_{sg,g,r,prior}$ are respectively, the additional landings in a given ICES rectangle remaining open to fishing following cessation of fishing in the MPA rectangles, and the landings in that rectangle prior to the establishment of the MPAs. $\sum_{r=x}^{OPEN} L_{sg,g,r,prior}$ and $\sum_{r=y}^{MPA} L_{sg,g,r,prior}$ are the total landings prior to the establishment of the MPAs from all rectangles remaining open to fishing and all rectangles designated as part of an MPA respectively. Next, the additional effort by each gear in each rectangle ($E_{sg,g,r,add}$) remaining open to fishing required to take the additional landings of principal species targeted by the gear needs to calculated. This is easily done knowing the catch per unit effort (CPUE) of each species in question in each of the main gears in each rectangle ($CPUE_{sg,g,r}$) by:

$$E_{sg,g,r,add} = \frac{L_{sg,g,r,add}}{CPUE_{sg,g,r}} \qquad 3.3.1.1.2$$

Finally, as a result of the additional fishing effort in the rectangles remaining open to fishing required to make up the shortfall in landings of the principal targeted species of each gear resulting from the closure of the MPA rectangles to fishing, other species of commercial value will also be caught. The additional landings of these commercially important "bycatch" species from each rectangle open to fishing can be determined knowing the CPUE of each species in each main gear in each rectangle:

$$L_{s,g,r,add} = E_{sg,g,r,add}.Cpue_{s,g,r} \qquad 3.3.1.1.3$$

Note the change in the subscript, from *sg* to *s*, in the Landings and CPUE terms to denote that we are no longer considering the species that are the principal target species of each gear.

Figure 3.3.1.1.2 indicated that the landings of sole, plaice and saithe would be the most affected by closure of the 14 rectangles indicated in Figure 3.3.1.1.1C. The two flatfish are primarily taken by beam trawl (Chapter 8) and use of this gear in the proposed MPA rectangles was higher than average across the North Sea. Saithe are primarily landed from otter trawlers targeting fish, and presumably effort by this quite specific fishery would be affected the most out of all otter trawling activity (Chapter 8). The results of the effort displacement models applied to beam trawl targeting sole and otter trawl targeting saithe are summarised in Table 3.3.1.1.1. The model output indicated that just these changes to the beam trawl and otter trawl fishery, driven by the targeting of sole and saithe respectively, were sufficient to make up the deficits in landings that would normally have been taken in the MPA rectangles for all species. In fact the data suggest that high levels of discarding (as high as 100% of landings) of cod, haddock, plaice and whiting may result from these changes. However, we have no way of assessing how excessive such discard levels may be, since we have no explicit data detailing discard levels that would have been associated with the landings of these species had they been taken as normal from the MPA rectangles. Figure 3.3.1.1.10 compares the spatial distributions of beam trawl and otter effort before and after the closure of the 14 ICES rectangles to form the MPAs. The maps show the redistributed effort predicted by the model. Figure 3.3.1.1.11 presents similar data for otter trawl effort directed at Nephrops and seine net, but in this instance, the MPAs are simply overlaid on the original effort patterns as the model suggested that little redistribution of effort by these gears would have been necessary.

| Main Gear Category | Beam Trawl | | Otter Trawl (Fish) | |
|---|---|---|---|---|
| Principal Target Species | Sole | | Saithe | |
| Time Period | 1997-2000 | 2001-2004 | 1997-2000 | 2001-2004 |
| Landings from rectangles designated as MPAs (t) | 4,628 | 4,331 | 6,584 | 14,620 |
| Effort expended in MPA rectangles to take landings (h) | 199,416 | 189,067 | 42,467 | 40,312 |
| Additional effort in "Open" rectangles to make up landings deficit (h) | 231,213 | 211,205 | 97,405 | 155,127 |
| Change in fishing effort over whole North Sea (h) | +31,797 | +22,138 | +54,939 | +114,815 |
| Effort in "Open" rectangles prior to MPA management (h) | 801,901 | 647,316 | 1,160,913 | 850,006 |
| Effort in "Open" rectangles after MPA management (h) | 1,033,115 | 858,521 | 1,258,318 | 1,005,133 |
| Percentage change in effort in "Open" rectangles | 28.8% | 32.6% | 8.4% | 18.3% |
| Percentage change in effort across the whole North Sea | 3.2% | 2.6% | 4.6% | 12.9% |

Table 3.3.1.1.1. Results of applying the effort displacement model to beam trawlers targeting sole and otter trawlers targeting saithe to determine the effects of closing 14 ICES rectangles to conserve species diversity in the demersal fish community.

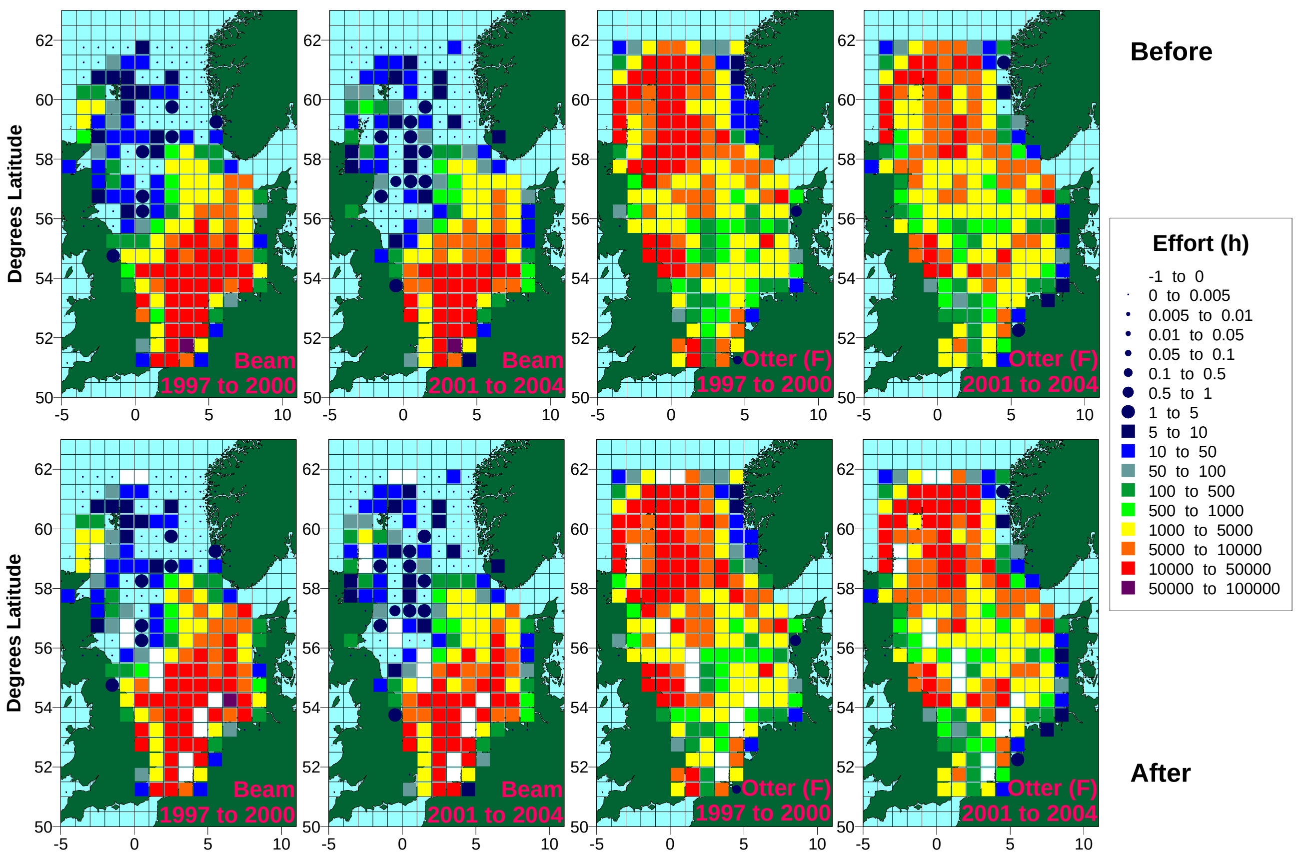


Figure 3.3.1.1.10. Spatial distributions of beam trawl and otter trawl effort directed at fish in two time periods, 1997 to 2000 and 2001 to 2004, before and after the designation of 14 ICES rectangles as MPAs designed to protect areas of high groundfish species diversity. After closure of the MPAs, effort is redistributed to make up the deficits of the two principal target species, sole and saithe, most affected by the locations of the MPAs.

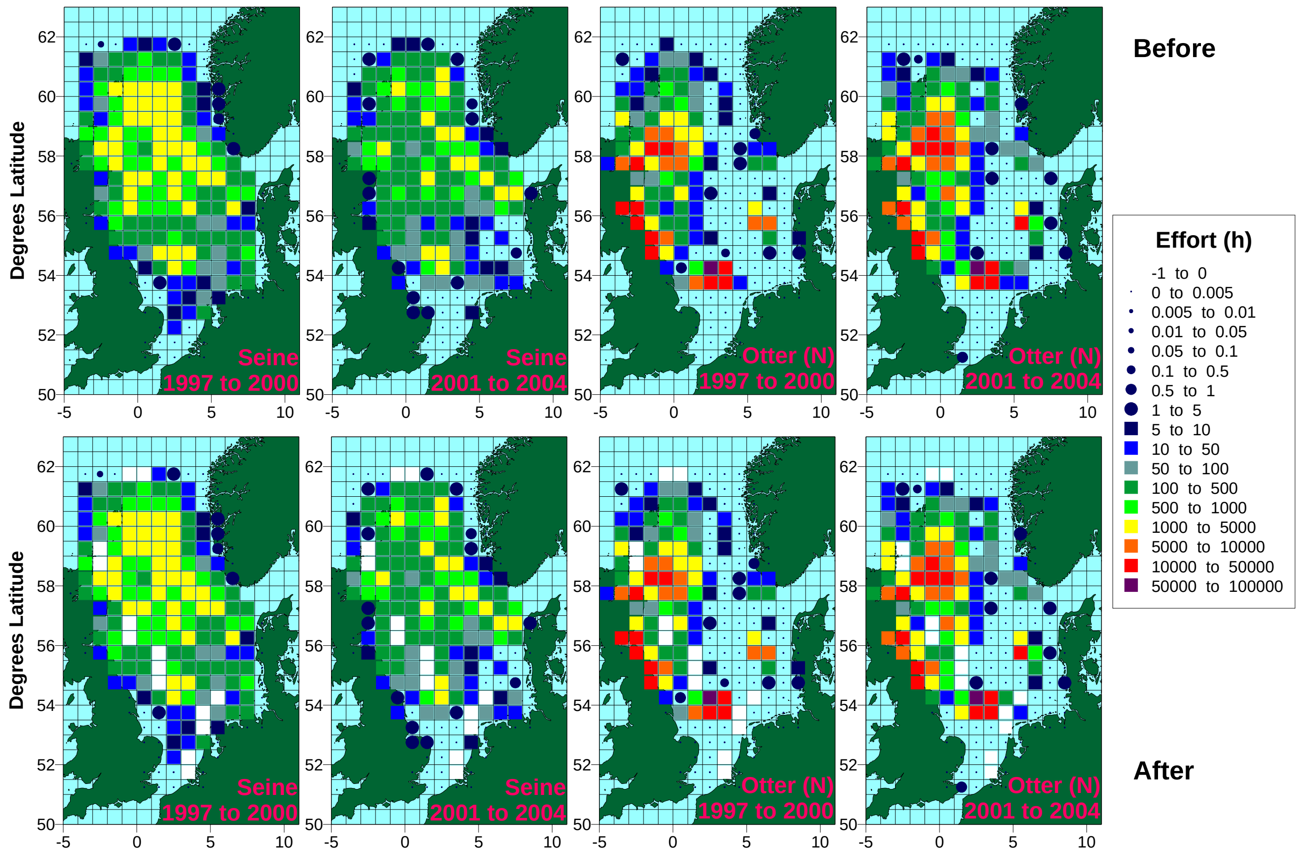


Figure 3.3.1.1.11. Spatial distributions of otter trawl effort directed at Nephrops and seine net effort in two time periods, 1997 to 2000 and 2001 to 2004, before and after the designation of 14 ICES rectangles as MPAs designed to protect areas of high groundfish species diversity.

In Chapter 8 (section 8.7) we present a benthic invertebrate disturbance model. This model estimates the percentage of benthic invertebrates that are killed each year in each ICES rectangle as a result of the fishing effort expended by each main gear category. This model can now be applied to the effort distribution data presented in Figures 3.3.1.1.10 and 3.3.1.1.11 to examine what effect closing 14 ICES rectangles to conserve groundfish species diversity has on benthic invertebrate communities in the North Sea. Figure 8.7.4.5.2 presents the mortality rate data for the two time periods, 1997 to 2000 and 2001 to 2004, that we have considered here, so we start from this point. Prior to the designation of 14 ICES rectangles as MPAs, average benthic invertebrate mortality across the whole North Sea was 19.8%.$y^{-1}$ between 1997 and 2000 and 16.5%.$y^{-1}$ over the period 2001 to 2004. These average values were based on the individual rectangle mortality rates weighted by rectangle area. If the 14 rectangles indicated in Figure 3.3.1.1.1 were closed to fishing and fishing effort was redistributed as indicated in Figures 3.3.1.1.10 and 3.3.1.1.11 to make up the deficit in landings that would normally have been taken in these MPAs, then average benthic mortality across the North Sea over these two periods predicted by the benthic disturbance model would be 20.1%.$y^{-1}$ and 17.1%.$y^{-1}$ respectively. Despite nearly 8% of the sea area of the North Sea being set aside as MPAs, in which all fishing activity was prohibited, overall impact on the benthic invertebrate communities across the entire North Sea increased in both time periods, by 1.2% in 1997 to 2000 and by 3.5% between 2001 and 2004, as a result of the introduction of closed area management. In both time periods, the overall increase in fishing effort required to make up the landings deficit, resulting from the considerable increase in effort outside the MPAs, combined with the fact that in many instances this increase in effort affected rectangles that had previously been relatively lightly fished, caused increased disturbance to benthic communities that outweighed any benefits gained from the introduction of the MPAs themselves. The actual spatial distributions of benthic annual mortality caused by fishing activity, before and after the introduction of the MPAs designed to conserve groundfish species diversity are shown in Figure 3.3.1.1.12.

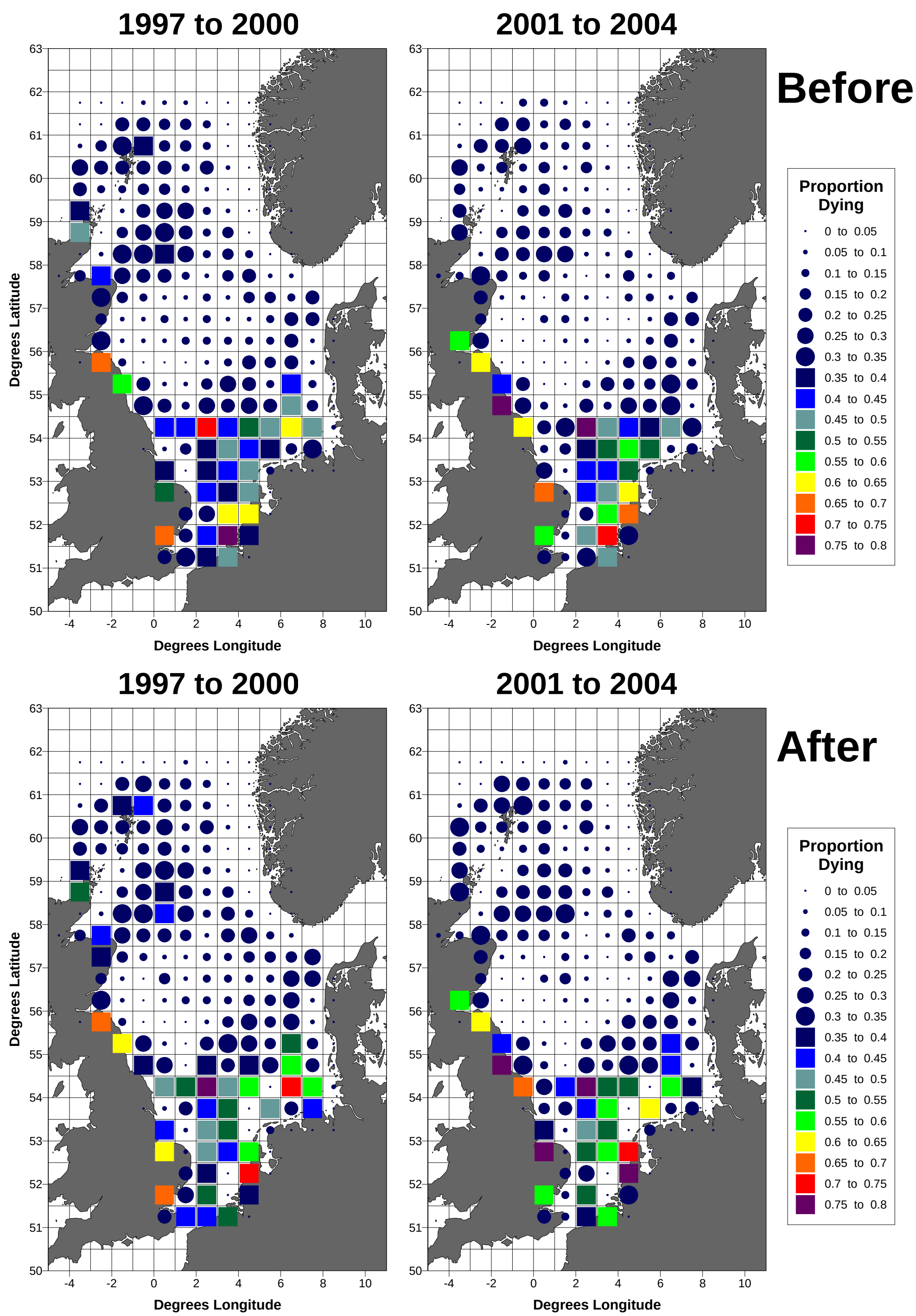


Figure 3.3.1.1.12. Spatial variation in the annual mortality of benthic invertebrates resulting from the combined fishing activity of beam trawlers, otter trawlers directed at fish, otter trawlers directed at *Nephrops*, and seine netters before and after the designation of 14 ICES rectangles as MPAs designed to conserve areas of high groundfish species diversity.

In conclusion, simply closing nearly 8% of the North Sea sea-area to protect regions of high species diversity in the groundfish community, without reducing TACs accordingly, results in an overall increase in fishing effort as fishermen attempt to make up the deficit in landings, normally taken from the MPA rectangles, by increasing fishing effort outside the closed areas. The increase in fishing effort in rectangles remaining open to fishing is substantial, resulting in increased impact on benthic invertebrate communities. Over the whole North Sea, the increased impact on the benthos in rectangles outside the closed areas outweighs any benefits gained from prohibiting fishing in the MPAs. For the fish community, the overall increase in fishing effort also has an impact, with indications of higher discard levels and greater impact on the non-target non-commercial species in the community. On balance, simply closing areas to fishing would appear to be of dubious value to the marine ecosystem. Real benefits may only be gained, making management through closing areas to fishing a viable tool for managers implementing an ecosystem approach to fisheries management, if at the same time TACs are also reduced accordingly. For the particular scenario on which this demonstration was based, Table 3.3.1.1.2 gives the reductions in TAC required.

| | 1997 - 2000 | | | 2001 - 2004 | | |
|---|---|---|---|---|---|---|
| | TAC | Reduction | % Reduction | TAC | Reduction | % Reduction |
| Cod | 117100.0 | 3709.1 | 3.2 | 38125.0 | 1017.2 | 2.7 |
| Had | 97637.5 | 2454.5 | 2.5 | 73441.3 | 2175.6 | 3.0 |
| Whiting | 52000.0 | 1041.4 | 2.0 | 23500.0 | 699.9 | 3.0 |
| Saithe | 101750.0 | 6588.9 | 6.5 | 144250.0 | 14621.9 | 10.1 |
| Sole | 20250.0 | 4655.5 | 23.0 | 16962.5 | 4385.8 | 25.9 |
| Plaice | 94250.0 | 10337.2 | 11.0 | 72312.5 | 9651.8 | 13.3 |
| *Nephrops* | 15700.0 | 71.6 | 0.5 | 17420.8 | 73.0 | 0.4 |

Table 3.3.1.1.2. Average annual TACs of each of the main commercial species over the periods 1997 to 2000 and 2001 to 2004, and absolute reduction and percentage reductions in TAC required in order to prevent redistribution of fishing effort following the establishment of MPAs covering 7.7% of the North Sea sea-area (14 ICES statistical rectangles) to protect areas of high species diversity in the groundfish community.

### 3.3.1.2. *MPAs and the Reduction of Fishing Disturbance to the Benthic Invertebrates*

Like the groundfish community, benthic invertebrate communities vary in species composition and structure across the North Sea with clear differences apparent between the northern and southern North Sea (Chapters 10 and 11). In order to safeguard all types of benthic invertebrate community, we selected groups of rectangles subject to high disturbance from fishing from across the entire North Sea, rather than concentrating potential MPA sites predominantly in the southern North Sea, where in general disturbance was highest. Distributions of benthic disturbance were shown in the preceding section (Figure 3.3.1.1.12). On the basis of these distributions we have selected the rectangles indicated in Figure 3.3.1.2.1 as potential MPAs with the objective of using these MPAs to reduce the overall impact of fishing activity on benthic communities throughout the North Sea. Rectangles where fishing mortality was amongst the highest within each local region of the North Sea were selected on the basis that this would have result in the greatest reduction in benthic mortality for the smallest area closed. The 15 ICES rectangles selected cover an area of 44,255km$^{-2}$, 7.3% of the sea-area of the North Sea.

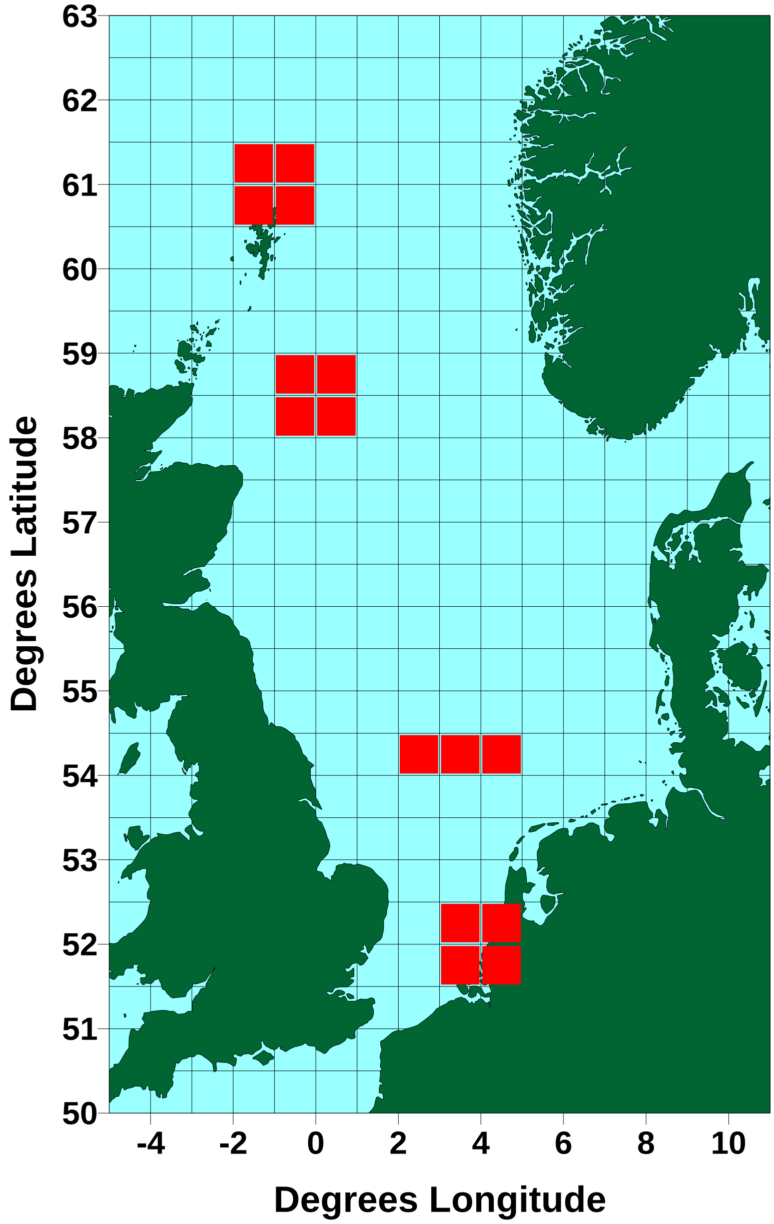


Figure 3.3.1.2.1. Location of four closed areas, including 15 ICES statistical rectangles covering 7.3% of the sea-area of the North Sea, designed to reduce the impact of fishing on the benthic invertebrate communities of the North Sea, in terms of reducing overall average annual benthic fishing mortality rates.

Since rectangles with amongst the highest benthic fishing mortality were selected for designation as MPAs, it is not surprising that the proportion of total North Sea landings of most species originating from the MPA rectangles was considerably higher than the proportion of the North Sea area set aside as MPAs. The location of the MPAs would appear to have the least affect on landings of haddock and saithe, but fishing activity for all other species, particularly Nephrops, seems likely to be severely affected by these MPAs (Figure 3.3.1.2.2A). With respect to the proportion of fishing effort by each of the main gear categories, all but seine fishing looks to be seriously affected by the location of the MPAs, and this is particularly the case with respect to otter trawl directed at Nephrops (Figure 3.3.1.2.2B).

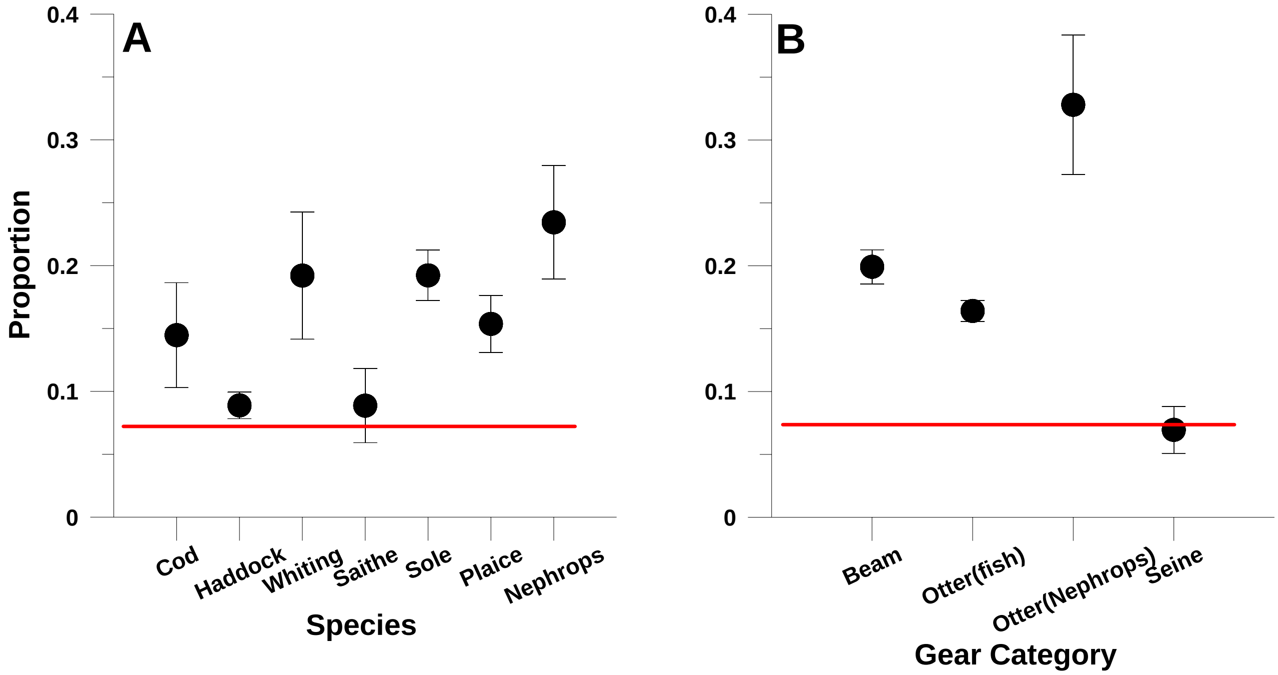


Figure 3.3.1.2.2. Mean proportion (±1 Std.Dev.) of total North Sea landings of each species (A) and fishing effort for each main gear category (B) reported from 15 ICES rectangles selected as MPAs to reduce the disturbance caused by fishing to benthic invertebrates over the period 1997 to 2004. Red lines indicate the expected fractions given the fraction of the area of the North Sea included within the proposed MPAs.

In the absence of concomitant reductions in TACs, to the extent indicated in Figure 3.3.1.2.2A, fishing effort is likely to be displaced to rectangles remaining open to fishing in order to make up the deficit in landings normally taken from within the MPAs. The effort displacement model described in section 3.3.1.1 was used to model the effort displacement required to make up the deficits in landings (Table 3.3.1.2.1). As suggested by Figure 3.3.1.2.2, it was only necessary to model effort displacement by three specific métiers to make up the deficits in landings normally taken in the closed areas; otter trawl targeting *Nephrops*, beam trawl targeting sole and otter trawl targeting whiting. In five out of six cases, overall effort declined following area closures and effort displacement, implying that the rectangles to which effort was displaced were more productive. For otter trawl targeting whiting and beam trawl targeting sole, the difference was negligible, but the reduction in overall effort by otter trawlers targeting *Nephrops* was quite marked. The *Nephrops* fishery is relatively restricted spatially and it just happened that the proposed MPA rectangles constituted a relatively high proportion of this area, and so included a high proportion of the landings and effort. But the rectangles closed to fishing tended to be the lower productivity rectangles (low CPUE). Closing these rectangles effectively forced fishing vessels into rectangles with higher CPUE. Trawlers utilized these lower CPUE rectangles because the gadoid bycatch was high and was commercially important to these vessels. Following displacement to rectangles outside the designated MPAs, CPU of *Nephrops* was higher, resulting in a substantial net decrease in effort required to make up the *Nephrops* deficit. However, the bycatch of gadoid species was approximately halved. With respect to whiting, the bycatch taken by *Nephrops* trawlers made up a substantial fraction of the whiting taken in the MPA rectangles. As a result, following redistribution of effort by beam trawlers and otter trawlers targeting whiting, the deficit of whiting landings previously taken in the MPA rectangles was not entirely made up. However, any attempt to rectify this resulted in substantial over-catches of saithe, cod and haddock.

| Main Gear Category | Otter Trawl (Nephrops) | | Beam Trawl | | Otter trawl (Fish) | |
|---|---|---|---|---|---|---|
| Principal Target Species | Nephrops | | Sole | | Whiting | |
| Time Period | 1997-2000 | 2001-2004 | 1997-2000 | 2001-2004 | 1997-2000 | 2001-2004 |
| Landings from rectangles designated as MPAs (t) | 1,782 | 3,273 | 3,816 | 3,484 | 2,729 | 1,594 |
| Effort expended in MPA rectangles to take landings (h) | 106,249 | 159,227 | 199,404 | 166,467 | 200,195 | 143,997 |
| Additional effort in "Open" rectangles to make up landings deficit (h) | 51,375 | 100,657 | 182,173 | 166,008 | 166,020 | 163,168 |
| Change in fishing effort over whole North Sea (h) | -54,874 | -58,570 | -17,231 | -459 | -34,175 | +19,171 |
| Effort in "Open" rectangles prior to MPA management (h) | 245,072 | 278,664 | 801,913 | 669,916 | 1,003,185 | 746,321 |
| Effort in "Open" rectangles after MPA management (h) | 296,447 | 379,321 | 984,086 | 835,924 | 1,169,205 | 909,489 |
| Percentage change in effort in "Open" rectangles | 21.0% | 36.1% | 22.7% | 24.8% | 16.5% | 21.9% |
| Percentage change in effort across the whole North Sea | -15.6% | -13.4% | -1.7% | -0.1% | -2.8% | +2.2% |

Table 3.3.1.2.1. Results of applying the effort displacement model to otter trawlers targeting *Nephrops*, beam trawlers targeting sole, and otter trawlers targeting whiting to determine the effects of closing 15 ICES rectangles to reduce the impact of fishing on benthic invertebrate communities.

Figures 3.3.1.2.3 and 3.3.1.2.4 compare maps of spatial variation in fishing effort before and after the establishment of the MPAs designed to reduce the impact of fishing on the benthos. The benthic invertebrate effort model was then run using these effort distributions to assess the effectiveness of these closed areas as a mechanism to reduce the impact of fishing on benthic communities. As we have already discussed in the previous section, prior to the introduction of any closed areas, benthic invertebrate fishing mortality was on average 19.8%.y$^{-1}$ over the period 1997 to 2000 and 16.5%.y$^{-1}$ between 2001 and 2004. After closing 15 ICES rectangles, 7.3% of the total sea-area of the North Sea, the model suggests that these overall mortality rates would have declined to 19.1%.y$^{-1}$ and 16.3%.y$^{-1}$, 3.8% and 1.7% reductions in the pre-closed area annual mortality rates over the two time periods respectively. Spatial plots of the distributions of benthic invertebrate fishing mortality in both time periods, before and after the introduction of MPAs, are shown in Figure 3.3.1.2.5.

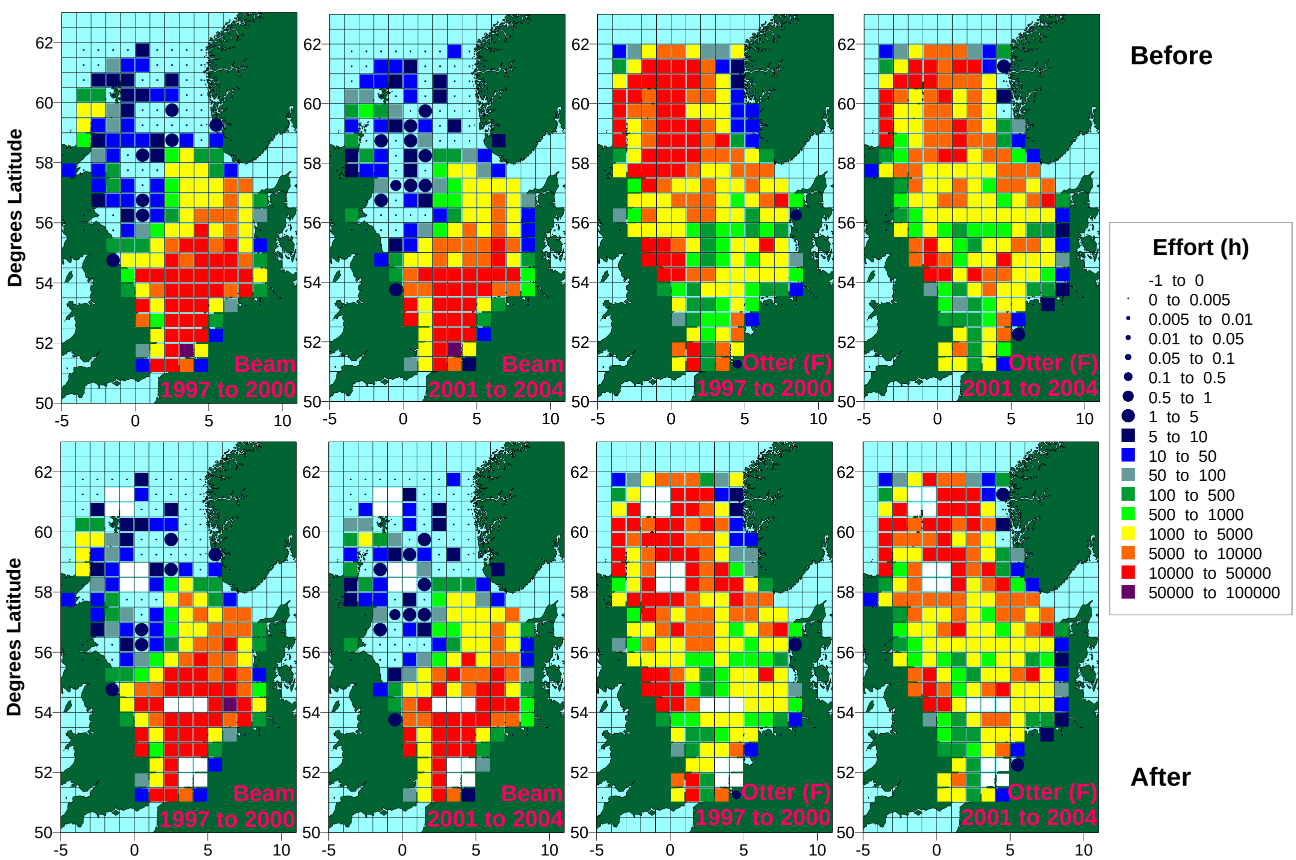


Figure 3.3.1.2.3. Spatial distributions of beam trawl and otter trawl effort directed at fish in two time periods, 1997 to 2000 and 2001 to 2004, before and after the designation of 15 ICES rectangles as MPAs designed to reduce the impact of fishing on North Sea benthic communities. After closure of the MPAs, effort is redistributed to make up the deficits of the two principal target species, sole and whiting, most affected by the locations of the MPAs.

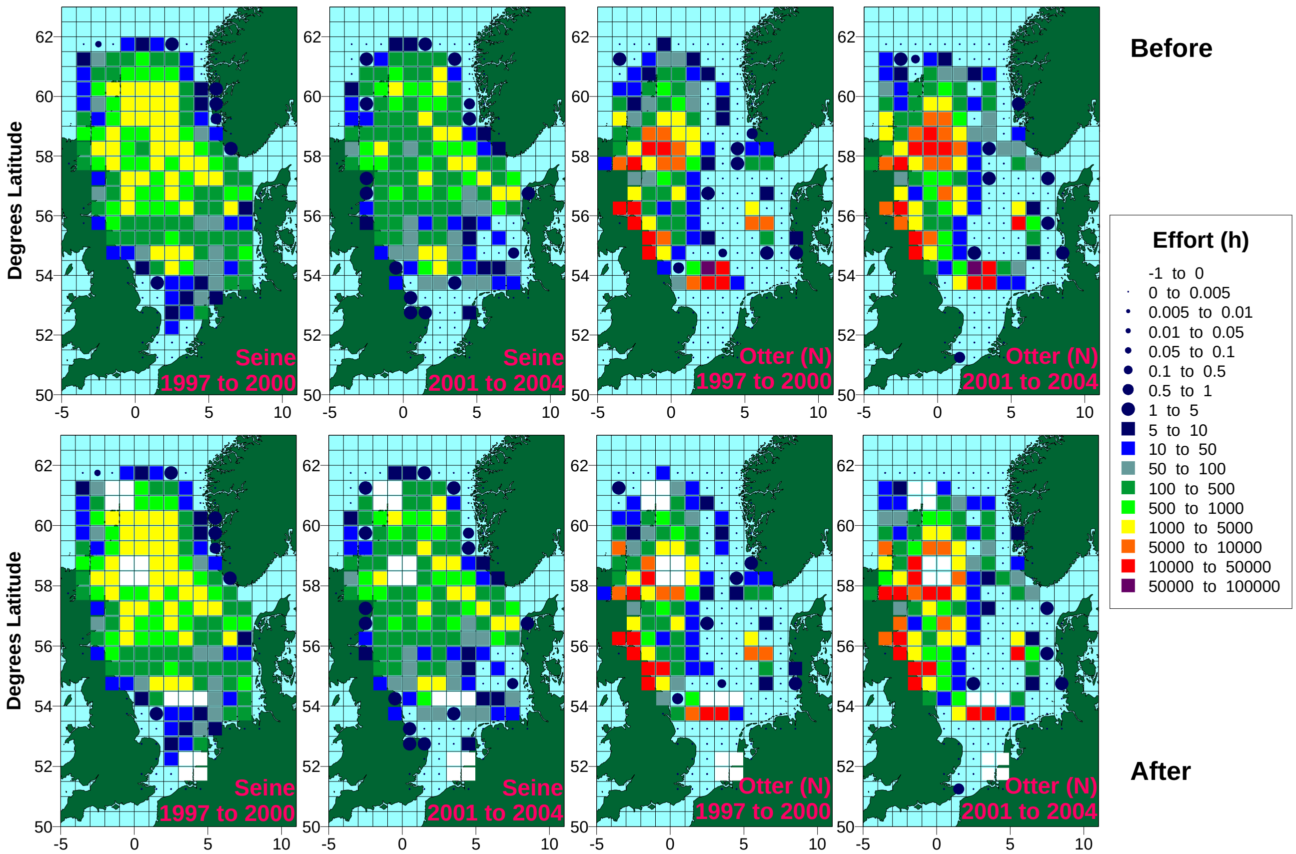


Figure 3.3.1.2.4. Spatial distributions of otter trawl effort directed at Nephrops and seine net effort in two time periods, 1997 to 2000 and 2001 to 2004, before and after the designation of 15 ICES rectangles as MPAs designed to reduce the impact of fishing on North Sea benthic communities. After closure of the MPAs, otter trawl effort is redistributed to make up the deficits of the principal target species, *Nephrops*, most affected by the locations of the MPAs. Redistribution of Seine effort was deemed unnecessary.

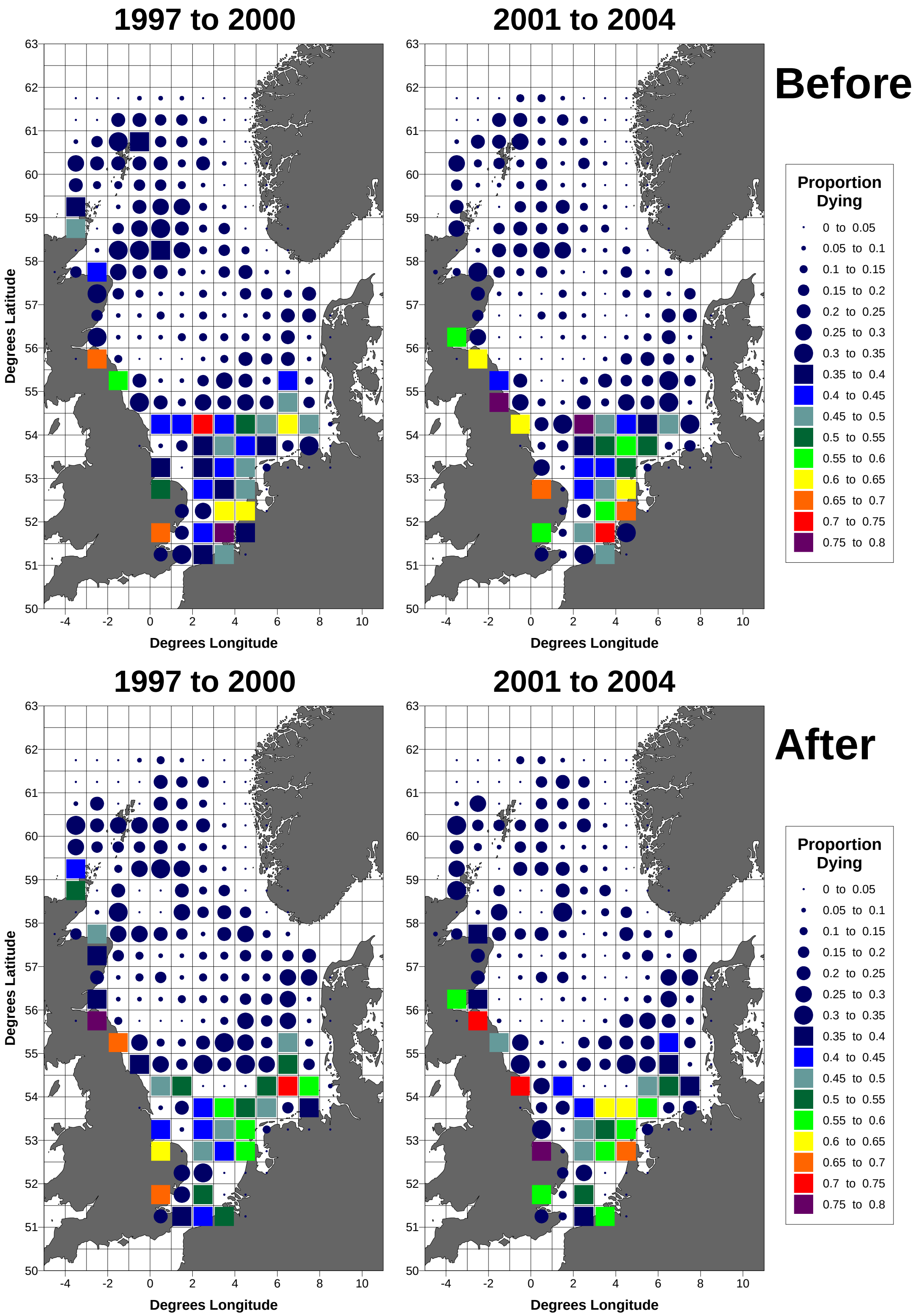


Figure 3.3.1.2.5. Spatial variation in the annual mortality of benthic invertebrates resulting from the combined fishing activity of beam trawlers, otter trawlers directed at fish, otter trawlers directed at *Nephrops*, and seine netters before and after the designation of 15 ICES rectangles as MPAs designed to reduce the overall impact of fishing on the benthic invertebrate communities of the North Sea.

Considering that approximately 7.3% of the North Sea was closed to achieve 3.8% and 1.7% reductions in overall North Sea benthic fishing mortality, closed area management with no other supporting action appears to be a relatively inefficient means of addressing such global scale ecological objectives. It is perhaps through the sheer chance that the rectangles selected for closure tended to result in the displacement of fishing activity to areas with higher CPUE, resulting, in most instances, in a reduction in overall effort, that any benefit in terms of reduced benthic fishing mortality was realised at all. The protocol described here can be used to examine the benefits of different closed area selections. In the scenario examined here we targeted rectangles with the highest benthic mortality rates, those with the highest levels of fishing effort. The relationship between fishing effort and benthic mortality is asymptotic in shape, not linear. Careful selection of rectangles could be employed so as to ensure that effort is displaced to areas higher up such curves. This should help to ensure that gains, in terms of reduced mortality, in closed areas always outweigh losses associated with increased mortality in areas to which effort is displaced. Once again, however, it seems that closed area management without associated reductions in TAC is relatively ineffective in addressing ecological objectives at the scale of the whole North Sea; the impact from fishing effort displaced throughout the remainder of the North Sea remaining open to fishing tends to negate much of the benefit achieved from the closed areas themselves. For the particular scenario on which this demonstration is based, Table 3.3.1.2.2 gives the reductions in TAC required. The combination of closing the 15 ICES rectangles along with the TAC reductions shown in Table 3.3.1.2.2, thereby assuming no effort redistribution, was examined using the benthic disturbance model. Under this set of conditions overall benthic invertebrate fishing mortality across the whole North Sea was reduced to 16.6%.$y^{-1}$ over the period 1997 to 2000 and 13.7%.$y^{-1}$ between 2001 and 2004; reductions in fishing mortality in each time period of 16.2% and 17.4% respectively.

| | 1997 - 2000 | | | 2001 - 2004 | | |
|---|---|---|---|---|---|---|
| | TAC | Reduction | % Reduction | TAC | Reduction | % Reduction |
| Cod | 117100 | 11067 | 9.5 | 38125 | 4873 | 12.8 |
| Had | 97638 | 7092 | 7.3 | 73441 | 4817 | 6.6 |
| Whiting | 52000 | 4363 | 8.4 | 23500 | 3243 | 13.8 |
| Saithe | 101750 | 6592 | 6.5 | 144250 | 9063 | 6.3 |
| Sole | 20250 | 3904 | 19.3 | 16963 | 3585 | 21.1 |
| Plaice | 94250 | 12780 | 13.6 | 72313 | 15168 | 21.0 |
| *Nephrops* | 15700 | 3116 | 19.8 | 17421 | 4121 | 23.7 |

Table 3.3.1.2.2 Average annual TACs of each of the main commercial species over the periods 1997 to 2000 and 2001 to 2004, and absolute reduction and percentage reductions in TAC required in order to prevent redistribution of fishing effort following the establishment of MPAs covering 7.7% of the North Sea sea-area (14 ICES statistical rectangles) to protect areas of high species diversity in the groundfish community.

### 3.3.2. Concluding Comments

We have demonstrated how the data sets compiled during the MAFCONS project, in conjunction with disturbance and effort redistribution models that we have developed, can be used to aid the selection of closed areas to achieve global scale ecological objectives for management in the North Sea. Our models are still at a preliminary stage. The benthic disturbance model can, for example, be improved to take account of mobility in benthic fauna. The effort redistribution model is currently very simple in concept, only utilizing information concerning spatial variation in landings, effort and landings per unit effort. This model can certainly be enhanced to take account of many additional factors, such as distance to home port, fuel costs, labour costs, fish market values, etc, all of which are bound to influence the decision processes of skippers faced with area closures on their favoured fishing grounds. For both the disturbance and effort redistribution models as they currently stand, accurate spatially referenced landings and effort data are essential. Already our data sets are becoming dated, and our ability as scientists to access the data necessary to keep these databases current seems limited. To provide relevant advice into the future, advice that reflects the prevailing situation, it is essential that the appropriate data are easily accessible. We return to this issue in Chapter 5.

Several points emerge from our two initial demonstrations of the MAFCONS management protocol. Firstly, as the benthic mortality scenario illustrated, even with unrestricted effort displacement, it may not always prove possible to make up the deficits in landings normally taken in the closed areas, at least not without considerable extra effort, and associated with markedly increased discarding. This will certainly have economic implications for the fishing industry. Secondly, an ecosystem approach to management will need to address numerous ecological objectives simultaneously. Considerable care will be required to ensure that MPAs established to address one specific ecological objective do not result in effort displacement that has detrimental impacts on other aspects of the marine ecosystem. Finally, closed area management alone may not be as effective a management tool to address North Sea-wide ecological objectives as many would hope. Effort displacement is a serious issue and the ecological consequences of increased effort in areas outside the MPAs may, over the North Sea as a whole, undermine much of the benefits gained from the closed areas themselves. However, MPAs combined with TAC reductions to reduce the need for effort displacement appears to hold considerable promise. Major steps forward towards the implementation of an ecosystem approach to management may be achieved through the combination of these two actions. The protocol we have started to develop here could provide scientists with exactly the tool needed to ensure that the advice scientists provide is adequate to support managers taking these steps.

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