5. Important Issues

5.1. The Need for Fishing Effort Data

An ecosystem approach to fisheries management is not about managing the marine ecosystem, it is about managing man's activities that adversely affect the ecosystem and threaten its capacity to continue to provide the ecosystem goods and services that we currently enjoy long into the future. This is best summed up in the Pressure-State-Response (PSR) framework for an indicator based ecosystem approach to management (Figure 5.1). Changes in "Pressure", reflected by variation in the pressure indicator, bring about changes in "State", reflected by changes in the state indicator. Management responds to this change in state, and this "Response" is reflected by changes in the response indicator. If the response is effective, it should illicit the anticipated change in the pressure indicator. The resulting change in pressure should then bring about the desired change in state. This simple concept absolutely encapsulates what we can aspire to as custodians of the world's marine natural resources. Our knowledge and our control over all the processes that structure and influence marine ecosystems is simply far too incomplete for us currently to consider that we can manage the marine ecosystem, and this may always be the case. The best that we can do therefore is to try and understand how man's activities impinge on marine ecosystems and to manage these activities so that detrimental impacts are kept to a minimum.

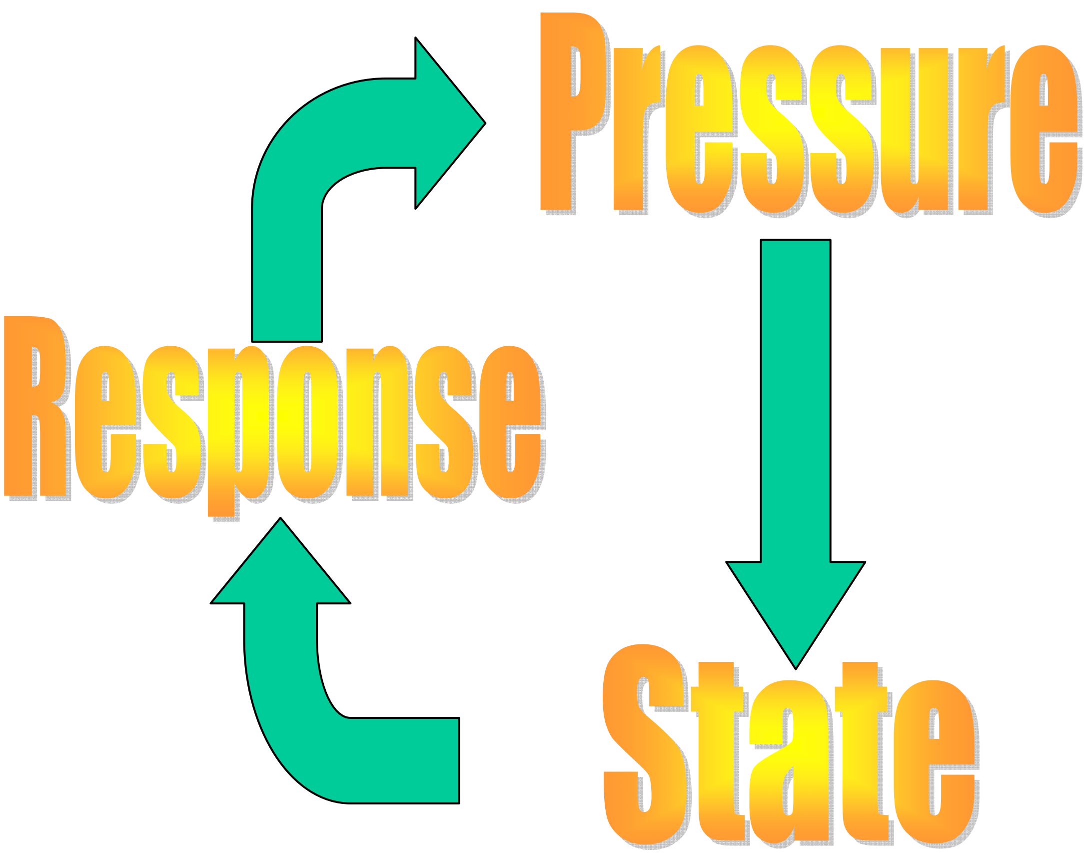

Figure 5.1. Representation of the Pressure-State-Response ecosystem approach to management framework.

Within such a framework it becomes abundantly clear that the relationship between “Pressure” and “State” is critical. Essentially, scientists providing the advice necessary to implement an ecosystem approach to management absolutely have to understand how man’s activities work to effect changes in the state of the system. If these relationships are poorly understood, then advice as to the correct “Response” will always be based on weak foundations. If this relationship is to be properly understood then accurate data describing the pressure is required. Of all man’s activities in the North Sea, fishing is the single activity that has the greatest impact on many diverse components of the marine ecosystem, from seabed habitats, through the benthic invertebrate communities and on up to the fish communities that constitute the resource being exploited. The need for accurate data describing the fishing “Pressure” is therefore of paramount importance to the successful implementation of an ecosystem approach to management. The need for such information has been amply demonstrated in the preceding four chapters of this report, and the problems faced by scientists in accessing such data are evident in the analyses presented in Chapters 8 and 12.

The traditional single species approach to fisheries management has evolved over several decades. Landings data collected through each nation’s market sampling programme provide the essential data for the stock assessment process. These data are analysed to provide estimates of fishing mortality for each individual stock. These mortality estimates provide the indicators of “Pressure” imposed by the fishing activity on each stock and are a vital component of the population dynamics models that form the basis of each assessment. These models produce estimates of stock abundance, the indicators of “State”, for each of the assessed stocks. Based on these estimates of the “State” of each stock, managers decide what level of catch can be sustained, and set TACs accordingly. These TACs therefore effectively provide the indicators of “Response” by management. In this example of the application of the PSR framework to a fisheries management context we argue that landings data provide the indicators of “Pressure”. Essentially market sampling schemes provide an indication of the numbers of fish landed, ie directly killed by the fishing activity. But on their own, market sampling data provide only a poor indication of the actual total numbers of fish being killed. Many fish are killed and landed illegally, or discarded at sea. Only by combining data collected by market sampling with information provided by discard observer schemes and fisheries protection and enforcement agencies, can estimates of the true numbers of fish being killed be determined that are adequate for the stock assessment and management process. It is clear from this discussion that in the application of the PSR framework to fisheries management, huge resources are expended in obtaining the necessary “Pressure” indicator data. The successful implementation of an ecosystem approach to fisheries management requires similar levels of effort to be directed towards obtaining accurate indicators of “Pressure” on other components of marine ecosystems. Currently such data are not available and there is little to suggest any major commitment to rectifying this situation.

For many non-target fish species, some indication of the impact of fishing on their populations may be obtained from discard observer schemes. But currently in the North Sea, only a small percentage of fishing trips carry an observer. Sampling effort is therefore low and any indicators of “Pressure” derived from such data will carry low levels of precision. For benthic invertebrates and seabed habitats, even data such as these are scarce or absent. For these components of the marine ecosystem of the North Sea it seems inconceivable that we will ever be able to directly assess the impact of fishing in terms of monitoring the numbers of non-target fish and benthic invertebrates killed, or the amount of seabed habitat altered, by fishing. In these instances, the most profitable approach lies in modelling the impact of fishing based on data that quantify levels of fishing activity in an appropriate way. This approach is exemplified in Chapter 8. Fishing effort statistics provide exactly the sort of quantitative measure of fishing activity required for this purpose. Landings data cannot be used as a proxy for this information because, as we show in Chapter 12, landings and effort are so poorly related. There are several reasons for this, but principal among these is the fact that the relationship between landings and effort is heavily influenced by variation in stock abundance. The practices of discarding, inevitable in mixed fisheries situations regulated by catch limitation, and illegal landings further decouple the relationship between officially reported landings and fishing effort. In order to implement an ecosystem approach to management, scientists therefore have to have access to fishing effort data of a quality, consistency, and as comprehensive as the landings data currently used in traditional fisheries management.

Currently, access to reliable fishing effort statistics is extremely restricted. No co-ordinated international initiative, similar to the market sampling and discard observer schemes has been initiated. The data that are currently available have largely been compiled by two separate EC funded research projects; the Biodiversity project covering the period 1990 to 1995 (Jennings et al 1999) and the current MAFCONS project covering the period 1997 to 2004 (Chapter 8). For neither of these projects were the databases fully comprehensive since not all countries operating fisheries in the North Sea were included in the project consortia. In the MAFCONS project, missing effort data for the countries that were not part of the project were modelled. However, such modelled data are a poor substitute for the real thing. In compiling these databases, the lack of co-ordination between the different countries quickly became apparent, with different countries using different gear codes and recoding the data with different units (hours-fishing or days absent from port). In order to render the data from different sources compatible, gear codes had to be combined until a "lowest common denominator" between the different countries was found, with considerable information loss as a result. Ultimately data could only be compiled for a few major gear-type categories (eg beam trawl, otter trawl, etc). For two countries, it was necessary to model hours-fishing from days absent from port to make the units in data from different countries compatible. The fact that the recording of effort data by fishing skippers appears not to be mandatory also caused problems in this respect. Compilation of a single comprehensive database from such disparate sets of data can only be achieved with the loss of information and a reduction in data accuracy. If the development and implementation of an ecosystem approach to management is to be a serious proposition, then the provision of routine fishing effort statistics is an absolute necessity. The accurate logging of such data by fishing skippers must be considered to be mandatory, with all countries recording similar data in an identical format. Such data collection needs to be co-ordinated and managed centrally, with the data fully accessible to scientists whose role it is to provide advice in support of management.

The two databases that have been compiled to date hold data as the number of hours fishing by main gear category in each ICES rectangle in each year. Thus both the spatial and temporal resolution of the data is relatively low. With centralised co-ordination, the provision of effort data on much finer temporal resolution (ie, by month, or week) should be easily achievable. The provision of finer scale spatial resolution is more problematic, and in all likelihood is the more important. In modelling the impact of fishing activity on benthic invertebrate populations (ie fishing mortality within the benthos) in Chapter 8, it was evident that effort data at the spatial scale of the ICES statistical rectangle was insufficient. Data for the Dutch beam trawl fisheries were available at a 1x1NM scale (Rijnsdorp et al 1998; Piet et al 2000; In Press) that allowed a benthic invertebrate model to be developed (Chapter 8), but similar data were not readily available to the project for otter trawls. Thus in modelling the mortality caused by otter trawls, major assumptions had to be made that at present have not been verified. The provision of shoot and haul positions for each deployment of the gear by fishing skippers would help to provide such information, but for precise modelling of the impact of fishing on benthic invertebrates and seabed habitats, the spatial information provided by the Vessel Monitoring by Satellite (VMS) scheme would be invaluable. Currently not all fishing vessels are monitored by VMS, and positions are only recorded every two hours, which is barely adequate for scientific needs (Deng et al 2005). Again, the requirement for high quality fishing effort data in order to implement an ecosystem approach to fisheries management is such that the VMS scheme should be extended to include all fishing vessels fishing in European waters and the monitoring frequency should be markedly increased. Management of the database should continue to be centrally co-ordinated and the data made readily available to scientists working to support the ecosystem approach to management.

5.2. The Issue of Catchability

Accurate assessment of the abundance of organisms is without doubt one of, if not, the most critical aspect of any study of ecological processes. Problems arising from species- and size-related variation in the catchability of surveyed marine fauna in the sampling equipment used affect marine science over a wide range of different topics. The issue of catchability is therefore of fundamental importance, not only to the MAFCONS study, but to all marine science undertaken in support of an ecosystem approach to management.

In studies of predator-prey interactions, knowledge of predator abundance is essential if total predation loadings are to be assessed (Bax 1991; Hislop *et al.,* 1991; Pierce and Santos, 1996; Mills and Shenk, 1992; Wanless *et al*., 1998; Furness, 2002; Daunt *et al*., submitted). Likewise, without knowing the abundance of prey organisms, these predation loadings cannot be converted to prey mortality rates (Sterner, 1986; Greenstreet *et al*., 1997; Hebblewhite, 2005). Such interactions are summarised by the classical Lotka-Volterra type differential equations for two predator and prey interacting species (May, 1976). Quantitative analysis of tropho-dynamic rates in food webs simply expands this process to include all species in the "assemblage" with significant predator-prey interactions (Pimm, 1982; Bax, 1991; DeAngelis, 1992; Greenstreet *et al.,* 1997; Blanchard *et al*., 2002; Araujo *et al*., 2005). Estimation of natural mortality rates is a key aspect of the stock assessments that underpin fisheries management in the North Sea (ICES, 2005). Information for this is largely drawn from Multi-species Virtual Population Analysis (MSVPA) models. Predator diet information for the MSVPA model is largely invariable, being mainly reliant on two North Sea wide diet studies carried out in 1981 and 1991 (Daan, 1989; Hislop *et al*., 1997). Most of the variation in estimates of natural mortality is driven by alteration of predator and prey species and size class abundance variable values (ICES, 2006a).

As we move from traditional fisheries management towards an "ecosystem approach to management" (Hall and Mainprize, 2004; Frid *et al.,* 2005), such considerations will need to be extended to include interactions between fish predators and a larger number of prey species belonging to a wider variety of taxa. Furthermore, while fisheries management has tended to operate at relatively large spatial scales (most stock assessments are undertaken at the North Sea wide scale or larger (ICES, 2005), such spatial scales may be entirely inappropriate for populations of some of these prey species, requiring estimation of fish predator abundance at much finer spatial resolution. For example, a considerable body of evidence now suggests that fishing has had a detrimental impact on benthic habitats and invertebrate communities in the North Sea (reviewed in Jennings and Kaiser, 1998; Collie *et al.,* 2000; Clark and Frid, 2001; Johnson, 2002; Kaiser *et al.,* 2006). Since "Benthic Communities" is Issue 6 in the list of ten issues for which Ecological Quality Objectives

(EcoQOs) are intended to be set (Lanters *et al*., 1999), mitigation of the impact of fishing on benthic invertebrates is likely to be a feature of the developing ecosystem approach to fisheries management. However, due to their smaller size and relatively sedentary nature, the impacts of fishing on the benthic invertebrate community are generally considered at spatial scales considerably below that of the whole North Sea, and often smaller than single ICES statistical rectangles (Kaiser *et al*., 1998; Collie *et al*., 2000; Craeymeersch *et al.,* 2000; Jennings *et al*., 2001; Dinmore *et al*., 2003; Kaiser *et al*., 2006). Benthic invertebrates are also subject to mortality from many sources, including natural disturbance from storms and phytoplankton "die-offs" (Taylor *et al.,* 1985; Hall, 1994) and predation from fish (Duineveld and van Noort, 1986; Greenstreet *et al.,* 1997). Through altering the abundance of populations of fish predators that prey on benthic invertebrates, fishing is also likely to have indirectly affected benthic invertebrate population dynamics (Frid *et al.,* 1999; Frid and Clark, 2000). If the processes by which fishing affects benthic invertebrate communities are to be sufficiently well understood so as to provide the scientific advice required to achieve benthic community EcoQOs, then accurate estimates of spatial variation in fish predator abundance are necessary.

Of particular relevance to the MAFCONS project is the impact of fishing on the species diversity of fish and benthic invertebrate communities. Fishing activity in the North Sea has increased markedly over the course of the 20th century (Daan *et al.,* 1990; Greenstreet *et al.,* 1999a) and consideration of the effects of this on various attributes of the demersal fish community has generated much interest (Gislason and Rice, 1998; Jennings *et al.,* 1999a; 2002; Piet and Jennings, 2005). There is now a large body of evidence to suggest that species diversity has been adversely affected, particularly in the northern North Sea (Greenstreet and Hall, 1996; Greenstreet *et al.,* 1999b; Hall and Greenstreet, 1998; Greenstreet and Rogers, 2000; 2006). In the southern North Sea the situation is less clear cut. Long-term declines in diversity have been noted (e.g. Rijnsdorp *et al.,* 1996), but other studies have suggested that groundfish diversity may have increased over time (Rogers and Ellis, 2000; Piet and Jennings, 2005). Huston's (1994) dynamic equilibrium model suggests that diversity may both increase or decrease in response to increased disturbance depending on local productivity. Thus increasing diversity in the shallow, hydrographically mixed, southern North Sea, where primary productivity is greatest (Reid *et al.,* 1990), may still be an "adverse" response to increased fishing disturbance. Policy drivers such as the Convention on Biological Diversity (CBD), Annex V of the Convention Protection of the Marine Environment of the North-East Atlantic (OSPAR), and the EC Marine Directive all stress the importance of conserving biodiversity. Such a goal should therefore feature highly in any ecosystem approach to the management of natural resources in the North Sea.

The use of protected areas to achieve ecological and conservation objectives is explicitly mentioned in several of these policy drivers. Article 8 of the CBD, for example, suggests that a system of protected areas be established where special measures are taken to conserve biological diversity. The UK marine strategy documents ("Seas the Opportunity" and Charting Progress") and the EC Marine Strategy Directive both explicitly consider the use of Marine Protected Areas (MPAs) to achieve their goals of "clean, healthy, safe, productive and biologically diverse oceans and seas". Indeed, (MPAs) are widely considered as potentially one of the most useful tools available to managers tackling ecological objectives for marine ecosystems (Russ and Alcala, 1989; Roberts and Polunin, 1991; Dugan and Davis, 1993; Agardy, 1994; Lindeboom, 1995; Allison *et al.,* 1998; Lubchenco *et al.,* 2003; Hastings and Botsford, 2003; Micheli *et al.*, 2004). Most analyses of North Sea groundfish survey data to date have involved time series analyses, linking changes in the community over time to temporal variation in fishing activity to demonstrate fishing effects (see references cited above). However, the use of MPAs to conserve species diversity in the North Sea clearly requires detailed knowledge of spatial variation in this component of the marine ecosystem, making spatial analysis of these data and mapping of groundfish and benthic invertebrate species diversity now a priority for marine scientists.

Estimation of the abundance of organisms is fundamental to predator-prey studies and studies of biodiversity. Groundfish surveys have been carried out in the North Sea in some form or other since the early 1900's (Greenstreet *et al.,* 1999; Rjinsdorp *et al.,* 1996). Research surveys provide estimates of the abundance of each fish species sampled at any particular location. However, no trawl gear ever samples all the individuals present in the path of the net and catch rates of fish of different species and size in any given fishing gear vary considerably. Vertical distributions of many species vary with time of day affecting "availability" of fish to demersal trawl gears (Benoît and Swain, 2003; Casey and Myers, 1998; Korsbrekke and Nakken, 1999; Michalsen *et al.,* 1996). Different species of fish behave differently ahead of the trawl gear; some are herded into the path of the net by the action of the otter doors and trawl sweeps on the seabed stirring up a sediment cloud (Bublitz, 1996; Main and Sangster, 1981; Ramm and Xiao, 1995; Somerton, 2004), others show net avoidance behaviour (Main and Sangster, 1981). Variation in swimming endurance influences which individuals fall back into the net (Michalsen *et al.,* 1996; Wardle, 1983; Winger *et al.,* 1999; Winger *et al.,* 2000). Several factors influence the catch efficiency and selective properties of trawl gears, such as sweep length (Engås and Godø 1989), mesh size (Suuronen and Millar, 1992), net spread (Engås and Godø, 1989; Rose and Nunnallee, 1998; von Szalay and Somerton, 2005), trawl speed and duration (Ehrich and Stransky, 2001; Somerton and Weinberg, 2001; Weinberg *et al.,* 2002) and the size and type of trawl ground-gear (Main and Sangster, 1981; Ehrich, 1987; Engås *et al*., 1988; Engås and Godø, 1989; Walsh, 1992). Consequently, the catchability of particular species and sizes of fish varies markedly between different fishing gears, dependent upon the characteristics of the gear (Ehrich *et al*., 2004; Harley and Myers, 2001; Winger *et al.,* 2004). Therefore, all trawl surveys provide gear-biased perceptions of the actual abundance of different species and size-class at any particular location. In essence this means that any particular research trawl provides a biased sample of what is actually present in the path of the trawl. Thus, the Grande Overture Verticale otter trawl (GOV) used in the ICES quarter 1 and quarter 3 International Bottom Trawl Surveys (IBTSs) provides a GOV biased view of the fish community and likewise the 8m beam trawl fishing gear used in the quarter 3 Dutch Beam Trawl Survey (DBTS) provides an 8BT biased view of the fish community.

In the time series analyses mentioned above, the fact that the derived estimates of community species diversity were subject to such bias was not an important issue. The studies involved analysed data collected from the same areas each year, so bias would have remained relatively constant allowing trends to be discerned. The presence of bias should not therefore have unduly affected the conclusions drawn. However, this is not the case with respect to analyses directed towards the examination of spatial variation within the same data. The species composition of the groundfish assemblage of the North Sea varies markedly across the region, being dominated by roundfish species in the north and flatfish species in the south (Daan *et al.,* 1990). Catchabilities of roundfish are much higher than flatfish in otter trawl gears, like the GOV, while the converse is true in respect to beam trawls (Ehrich *et al*., 2004), hence the reason why otter trawling dominates commercial fishing activity in the northern North Sea, while beam trawling predominates in the south (Jennings *et al.,* 1999b). Spatial patterns of the species diversity of demersal fish community derived from the GOV and 8BT are therefore strongly gear dependent and differ considerably (Figure 5.2.1). Furthermore, neither may reflect the real spatial pattern in species diversity of the actual demersal fish community present across the North Sea. Without knowledge of the bias caused by variation in the catchability of the different sizes and species of fish in different research trawls, maps of the species diversity of the North Sea demersal fish community derived from spatial analysis of groundfish survey data may well prove to be miss-leading. Under such circumstances, their use to underpin advice on which to base closed area management would appear to be seriously flawed.

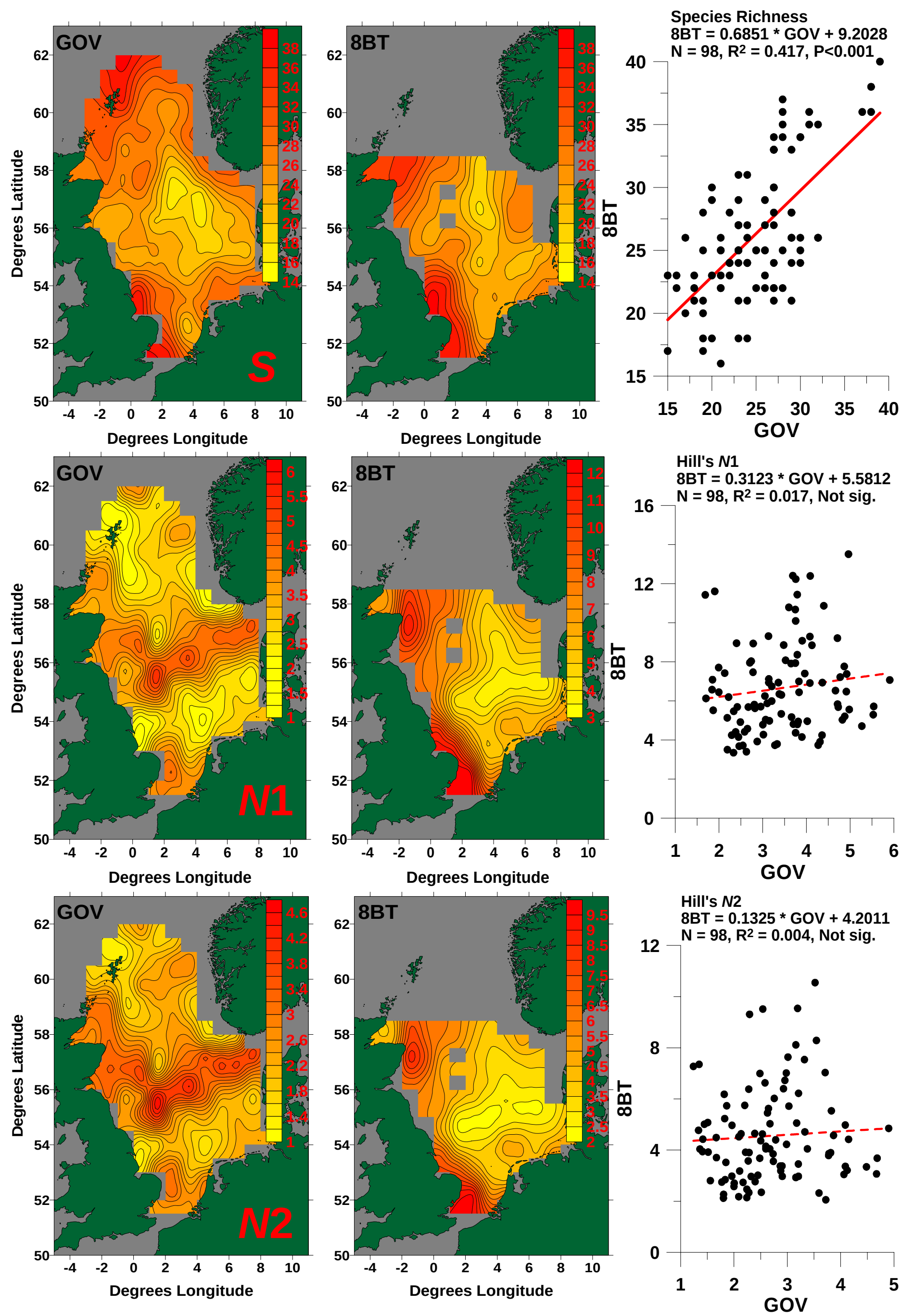

Figure 5.2.1. Spatial variation in species richness (S) and Hill's N_1 and N_2 indices of species diversity derived from the GOV (IBTS) and 8BT (DBTS) data sets. Correlations in index values obtained from each data set are also shown.

In order to estimate actual species densities at each location, survey trawl catch density estimates need to be converted to estimates of actual absolute density by taking into account the catchability of the fish involved in the particular gear employed (Harley and Myers, 2001). The issue of catchability in sampling gears is not just restricted to the estimation of fish abundance, determining the abundance of benthic invertebrates is subject to similar difficulties (Reiss et al 2006). Issues of species catchability in IBTS trawls have come to the fore in previous studies that attempted to estimate the total biomass of fish in the North Sea (Yang 1982; Daan *et al.,* 1990; Sparholt 1990). These studies equated groundfish survey catch rates of the main assessed commercial species with estimates of their biomass in the North Sea derived from the ICES stock assessment process. All the non-assessed species were then assigned to a "fish-type" group, each of which was headed up by one or more of the assessed species. Within each of the "fish-type" groups, biomass of the non-assessed species (B_{na}) was estimated by:

$$B_{na} = \frac{C_{na}}{C_a} \times B_a$$

where C_{na} and C_a are the catch rates of the non-assessed and assessed species in each group respectively and B_a is the biomass of the assessed species in the group.

During the MAFCONS study we have developed this approach further, utilising data collected in two surveys using very different research trawls, the GOV and the 8BT, to model variation in catchability, q, for each 1cm size class of every species sampled by the GOV trawl in the ICES quarter 3 IBTS (Fraser *et al.,* 2006 submitted; Chapter 9). Application of the catchability correction factors to the GOV sample data to derive estimates of true abundance had a profound effect on our interpretation of spatial variation of both the species diversity (Figure 5.2.2) and productivity (Figure 5.2.3) of the groundfish assemblage. Failure to take account of the catchability of different species and size classes of fish in the GOV produces a miss-leading impression of where species diversity and productivity hotspots are located. Any attempt to understand the processes that influence, structure and control these attributes of the fish community would therefore be entirely false. Add to this similar difficulties with respect to benthic organisms and it is soon apparent that our understanding how marine ecosystems operate may be seriously flawed. Assessing the catchability of marine organisms in the samplers used to assess their abundance is therefore a critical issue that needs serious and urgent attention. Failure to address this question could seriously compromise the scientific advice provided in support of an ecosystem approach to management at just about every level. If as a consequence management action consistently fails to produce the anticipated response so that managers lose faith in the process, this may eventually jeopardise the management approach itself. But ultimately, consistent ill-advised and inappropriate management may put the marine ecosystem at risk.

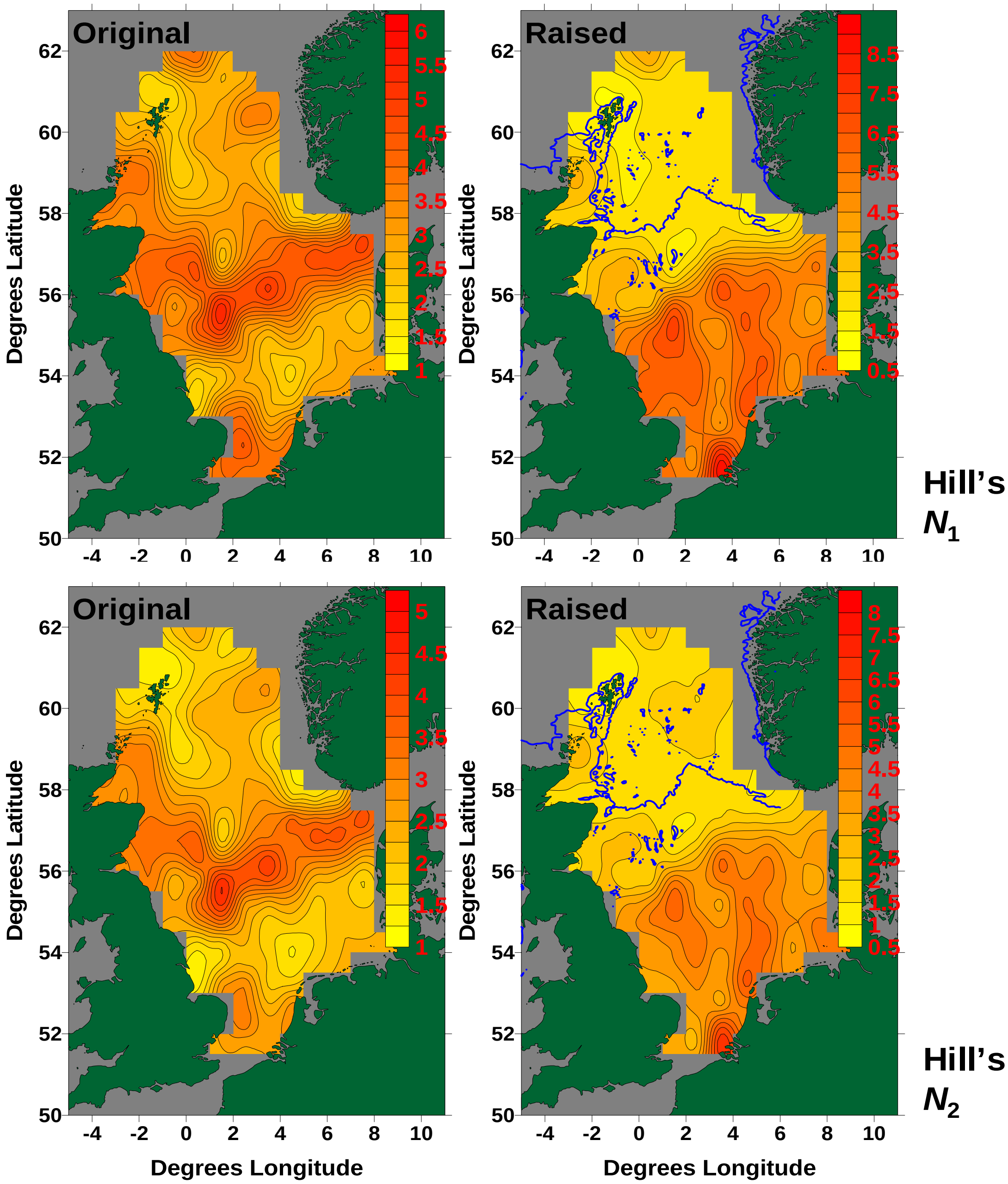

Figure 5.2.2. The effects of taking catchability into account on spatial variation in Hill's N_2 based on the IBTS GOV data-set. Original: based on the raw trawl data. Raised: based on the raw trawl data corrected for catchability.

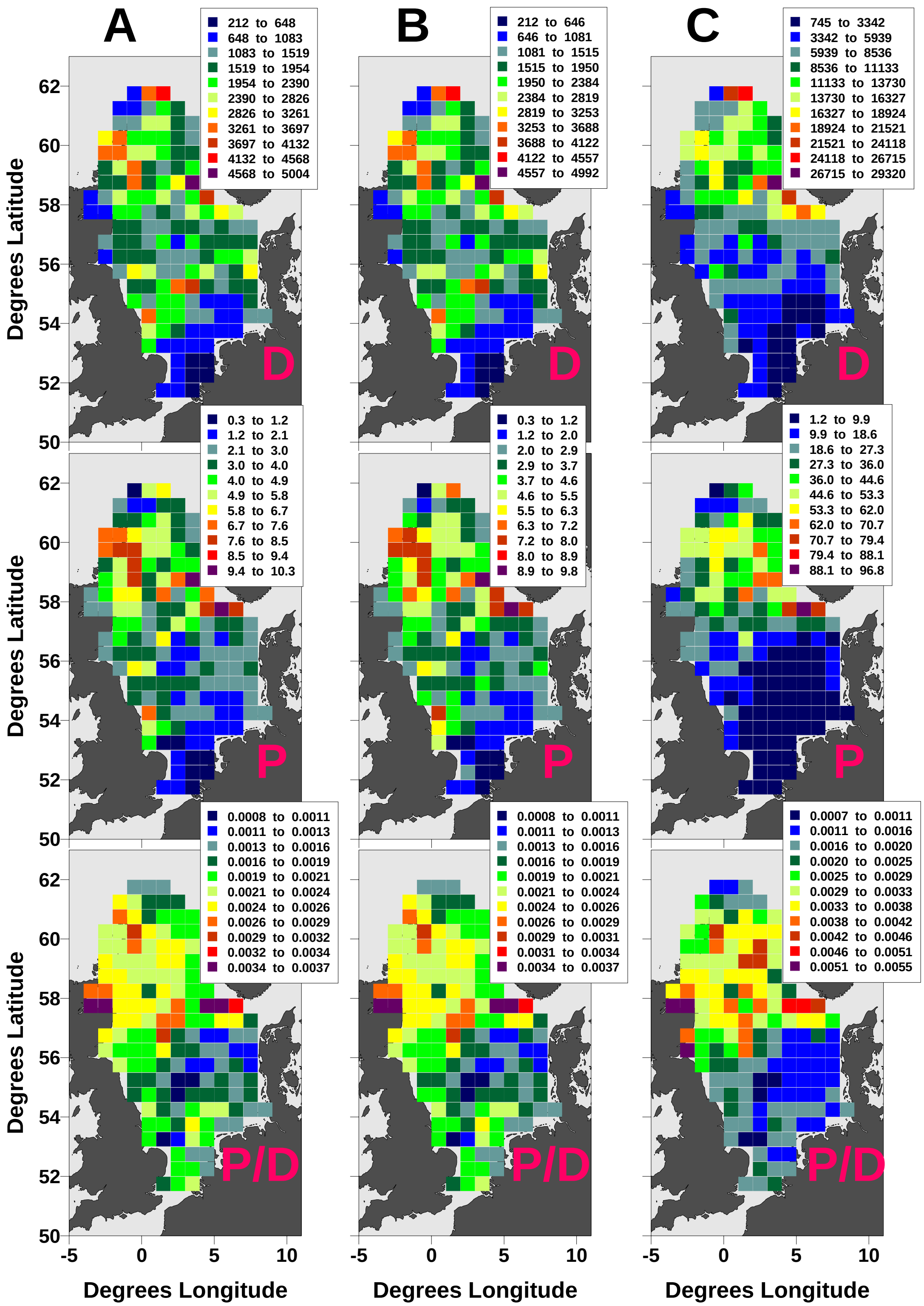

Figure 5.2.3. Spatial variation in the biomass density (D, $Kg.Km^{-2}$), growth production (P, $Kg.Km^{-2}.d^{-1}$), and production per unit biomass (P/D) of the demersal fish community. A: Based on arithmetic mean densities calculated across all 20 hauls in each rectangle on the raw uncorrected (for 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.

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