The Ecological Disturbance of Fishing in Demersal Fish and Benthic Invertebrate Communities

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Work Package *3*	Deliverable *3*	MAFCONS Report 2003:003
Managing **F**isheries to **Cons**erve Groundfish and Benthic Invertebrate Species Diversity (**MAFCONS** Project: EC project number Q5RS-2002-00856)		

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CONTENTS

1. INTRODUCTION

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

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

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

The main objective of this review is, therefore, to define the different sources of fisheries mortality in demersal fish and benthic invertebrate communities. Changes in habitat structure that occur following the passage of fishing gear are not implicitly considered at this stage, but the potential to develop disturbance indices that account for the implications of habitat change is discussed. Undoubtedly, levels of fisheries mortality experienced by a community will vary depending on a number of factors, including the type of fishing gear and power of the vessel being used, the target species of the vessel, and, particularly in the case of benthic invertebrates, the habitat type (Kaiser & Spencer, 1996b; Bergmann *et al.*, 2002; Thrush & Dayton, 2002). For example, it has been found that for each hour of beam trawl effort many more benthic invertebrates and flatfish will be killed than pelagic and demersal roundfish, in comparison with an equivalent hour of otter trawling (Philippart, 1998). As a major second aim, the review will consider the available measures of fishing effort (from the fisheries statistics) in the light of the potential to develop indices of ecological disturbance from them. Until now, indices of the disturbance caused by fishing have been derived directly from the fishing effort data with no weighting added for how factors such as gear type will cause variation in the mortality induced. These indices are usually represented as either a basic scale of the number of hours of fishing in a particular area, or, as the area of the seabed 'swept', or the volume of water trawled in a given time period (calculated by multiplying total effort hours trawling, by the area/volume swept per hour (average width of trawl (km)/volume of trawl (km^3) x distance per hour (km/hour)).

The overall objective of the development of this review is to provide the information necessary to establish what data are needed to develop indices of disturbance to demersal fish and benthic invertebrate communities on a North Sea Scale. This would allow for the prediction of the level of mortality experienced by a particular community given a combination of the number of hours fishing in that area by particular fleets. It is hoped that when the actual indices are developed it will be possible to map the spatial distribution of fishing effort broken down by fleet and gear types, in order to predict the actual ecological disturbance in that area. The available effort statistics will themselves be thoroughly examined to see whether more could be made of the information available, for example by incorporating information on vessel power as well as the hours actually spent fishing by each vessel. The community level response to a particular level of ecological disturbance following fishing activity, will then be based on the interaction of a number of factors, including the level of local productivity, the spatial extent of mortality to individual populations and the history of disturbance regimes in the locality. It is not the aim of this review to attempt to develop the overall models of community change following fishing. The implications of the various sources of fisheries mortality on both demersal fish and benthic invertebrate communities will however be examined in the final sections.

2. FISHING DISTURBANCE TO DEMERSAL FISH COMMUNITIES

2.1 Mortality of target stocks

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

Although the ICES assessments will allow the examination of the mortality of target species in the North Sea, data are only available at the scale of a sub-region, each of which cover many ICES statistical rectangles. If the aim is to devise indices of ecological disturbance at the smaller ICES rectangle scale it will be important to examine the individual landings for each country fishing in the sub-region as these data are provided at the scale of the rectangle. Using the landings data it may then be possible to determine the mean annual spatial distribution of landings for particular stocks on a North Sea scale. Some preliminary analyses are currently being undertaken at the FRS Marine Laboratory in Scotland to spatially map distribution of mortality of particular stocks based on the catch composition of the trips made by individual Scottish fleets in a given area (Liz Clarke, *pers. comm.*).

Clearly the data available to calculate fisheries disturbance of target stocks based on landings are of a high resolution. However, these data only provide information on one component of the total fisheries mortality that is experienced by the individual species. A proportion of the target species caught will never be landed and further mortality will be experienced by those individuals that escape the net (escapee mortality as described in Section 2.3). Large numbers of target species may be caught but subsequently discarded, either because they are below the minimum landing size (MLS) or because the vessel does not have a quota, or is over quota for that species. In some cases, animals that have been caught are later discarded if higher quality specimens are found taking the vessel over quota (a practise known as 'high-grading'). Fishermen have also been observed to discard landable target fish when there is no current market for that species (Cotter, 2003). It will therefore be important to consider fluctuations in marketability of target stocks for those years that disturbance indices will be derived.

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

At the same time it should be recognised that these numbers do not represent absolute mortality as it is possible that some animals will survive following discarding. Some field studies have measured discard mortality by holding fish for specific time periods after capture or by using tag and recapture methods, but it has been suggested that the accuracy of such measures is relatively limited (Davis, 2002). There are however data from experimental studies for some target species and it will be important to investigate whether there is enough information to add any level of survivorship to the discarded numbers when determining the disturbance indices (e.g. Van Beek *et al.*, 1990). Many factors may affect the level of survivorship in discards, including the total time spent on deck, time spent in the codend before hauling and individual species physiological responses to changes in environment (Davis, 2002). In deriving disturbance indices the resolution of data may only allow that survivorship estimates of discarded species vary by gear, fleet and species.

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

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

Considerable levels of mortality are experienced by some target fish species that are caught and discarded by fisheries targeting benthic invertebrate species. For example, Frid *et al*. (2000) suggest that on average, 26% of the catches (by weight) from fisheries in general are discarded, but that in shrimp/prawn fisheries discards are as much as 84% of the catch. However, Garthe *et al*. (1996) suggest that on a North Sea scale the amounts of discards of fish from invertebrate targeting fleets are low relative to those fleets targeting fish, simply because the fish targeting fleets expend much greater effort overall. However, as fisheries for stocks such as *Nephrops* have developed in recent years, it is important to consider the potential changes in overall discarding of fish that may occur as a result of increasing effort by these fleets.

For example, Evans *et al*. (1994) recorded discards of small Whiting in the Farne Deep *Nephrops* fishery, that were greater than the total catch of the targeted *Nephrops*. On average each vessel was discarding 11,000 undersized Whiting each day! There were also discards of other commercially important fish species and some non-target species and although these were in less significant quantities there are available data for some species. Revill & Holst (2004) report that the Brown shrimp fishery has long been associated with a high bycatch of juvenile fishes. A European discard study conducted during 1996-1997 estimated that over 900 million juvenile Plaice (*Pleuronectes platessa*) were discarded in this fishery in a single year (Van Marlen *et al.*, 1998). In a European Commission Report cited in Revill & Holst (2004), it was estimated that such levels of discarding might ultimately result in 7000-19000 tonnes of foregone plaice landings in the North Sea. This equates to 10-25% of the 1998 total allowable catch (TAC) for Plaice in the North Sea and would therefore mean that potentially, in the Brown shrimp fishery alone, an extra mortality of 10-25% of the TAC for plaice has been occurring each year. Clearly such levels of unaccounted mortality must be included in any realistic estimation of the ecological disturbance caused to the demersal fish community. It is also suggested that, albeit to a lesser extent, there a number of other commercially important fish species discarded by the Brown shrimp fishery, including Cod (*Gadhus morhua*) and Whiting (*Merlangius merlangus*) (Revill & Holst, 2004). It is not however clear whether there are significant discards of any non-target demersal fish such as the common dab. At the community level, it will obviously be important to consider the importance of the discarding levels of all these species in fisheries targeting benthic invertebrates, particularly as high proportions of the discarded mortality are juveniles (Evans *et al*., 1994; Bergmann *et al.*, 2002; Revill & Holst, 2004).

At the same time, it is important to note that technical measures are currently being introduced, or have already been introduced, to reduce the levels of discarding for some of these fisheries. In some cases these measures have been extremely successful, considerably cutting discarding rates of fish and benthos alike. For example, in trials conducted by Revill & Holst (2004) in the 1999 and 2000 brown shrimp fishing season, the use of a sieve net that directs larger animals such as fish and larger benthic invertebrates out of the trawl net, resulted in a 90% reduction in the retention of unwanted by-catch (both fish and benthic invertebrates), with only an 8% loss of the target species. Of the designs of sieve nets trialed this was the most successful, although all of the designs reduced by-catch by over 56%. In contrast to this however, Bergmann *et al*. (2002) found that although precautionary measures such as the use of square mesh panels were mandatory at the time of their study in the Clyde Sea *Nephrops* fleet, undersized commercial fish still accounted for up to 39% of the catch. Undoubtedly it will be important to try to establish both the success rates of the different technical measures, and when the measures have been or are going to be implemented. This will help to predict how the ecological disturbance of such fisheries should change as technical measures are established (See Section 4.3). Revill & Holst (2004) observe that sieve nets have been mandatory in the Danish brown shrimp fishery for many years and that all member states of the EU signed up to the implementation of such measures in January 2003.

2.2 Mortality of non-target stocks

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

In comparison to the bycatch of non-target benthic invertebrates (described in Section 3.2), the availability of data on non-target mortality in both the landed catch and discarded fraction is good, particularly in recent years. As described in Section 2.1, new EC regulations on Data Collection (EC Reg. 1639/2001) have made it mandatory for member states to collect discarding data on both targeted and non-targeted fish stocks. In theory there should be open access to recent data for the levels of discarding of non-target fish stocks from all countries with membership to the EC (J. Cotter, 2004, *pers. comm*.). However, examination of trends in the discarding of non-target species will be more difficult to access due to a lack of North Sea scale historical data. There is also much less known of the survivorship of discards in non-target species.

2.3 Mortality of demersal fish in the tow path of the gear

Since the mid 1980s research institutes in Scotland and Scandinavia have been conducting experiments that attempt to measure the survival rates of demersal roundfish that escape through the codend mesh during the fishing process (Wileman *et al*., 1999). Those animals that die following passage through the codend are a source of mortality from the fishing disturbance that is unaccounted for in any of the methods described above. The experiments that have been undertaken into escapee mortality attempt to quantify the proportion of those fish escaping that would die just from the physical injuries and stress induced in the trawling process. This is different to those animals that would suffer natural mortality through predation as a consequence of their weakened state. This latter element of mortality is considered here to be an element of the community level response to the fishing disturbance as described in Section 5.

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

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

Although these experiments provide important information on the proportion of fish that survive after passing through the trawl gear, the work has only been undertaken on a small number of species from the demersal fish community, in most cases only the commercially important roundfish species. For these species the work undertaken suggests that this level of mortality is insignificant at the scale of the stock. However, in determining the ecological disturbance of fishing it will be important to consider whether this additional source of mortality is significant at the local community scale. It will also be important to investigate whether there has been any work undertaken on the escapee survival rates of other targeted and non-targeted species.

3. FISHING DISTURBANCE TO BENTHIC INVERTEBRATE COMMUNITIES

3.1 Mortality of target stocks

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

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

Nephrops are exploited throughout their geographical range from Iceland to the Moroccan coast of the Mediterranean. They have been exploited commercially in the North Sea since the mid 1970s and there are important fishing grounds off the northeast coast of England in the Farne Deep and off the northeast coast of Scotland on the Fladden Ground (Evans *et al*., 1994; Marrs *et al*., 2000). Annual landings are around 60,000 tonnes and about one third of this is landed into Scotland.

TACs for *Nephrops* stocks have been imposed since the 1980s (Marrs *et al.*, 2000) and stock assessments are undertaken annually by the ICES *Nephrops* stock assessment group. Recent EC studies of the North Sea and Clyde Sea *Nephrops* stocks, have however expressed concern at the methods used to calculate the annual stock assessment. This is because the assessments rely on assumptions that are more suitable for finfish stocks, such as homogeneity of the stock, equal capture availability and a finfish behaviour model for redistribution of the stock following capture of part of it (Marrs *et al*., 2000; Marrs *et al*., 2002).

Nephrops have been shown to exhibit little homogeneity in their stock size and distribution as their population biology is closely linked to the sediment type and local hydrodynamics (Tuck *et al.*, 1997; Marrs *et al*., 2000; Bergmann *et al*., 2002). There are also inherent differences in the catchability of the different sexes and ages. Berried females and juveniles spend most of the time in the burrow, being much less vulnerable to capture than the more active adult males (I. Tuck & R.J.A. Atkinson, *pers. comm.*). A new EC study will be trying to resolve the relationship between effort and mortality in the North Sea Fladden Ground *Nephrops* fishery (Ian Tuck, *pers. comm*.). It is hoped that the findings of this study will help to improve the stock assessments and thus may provide a more ecologically robust input for target mortality of *Nephrops* in the invertebrate disturbance index.

Invertebrate fisheries are known to have extremely high discard rates, with the total discards far exceeding the weight of landings (Evans *et al.*, 1994). Much of this discarded material consists of the bycatch invertebrates and fish, but there are also substantial quantities of discarded target stock. These are the animals that are either under the minimum landing size (MLS) or those that are discarded because the vessel is over quota and/or better quality specimens above the MLS are found. It is also likely that there will be discards of target invertebrate stocks from vessels operating to target demersal fish. This is further complicated in fisheries that target for both *Nephrops* and gadoids such as Haddock and Whiting. Under the new EC Data Collection regulation, each member state should be collecting data on discarding rates of all target stocks including invertebrates in all operating fisheries (See Section 2.1). This may provide invaluable data on the mortality of target stocks not recorded in the landings data.

Of the targeted benthic invertebrate species, there has probably been the most work done on discard rates of *Nephrops norvegicus*. In order to account for the mortality of discarded animals, it is important to know both what proportion of a catch is usually discarded and what proportion of those animals discarded survive. Evans *et al*. (1994) studied discarding rates of *Nephrops* in the

Farne Deep fishery and found that on average 63.2% by weight (or 85.3% by numbers), of *Nephrops* caught were discarded. A study by Bergmann *et al*. (2002) in the Clyde Sea, does however suggest that discarding rates of *Nephrops* are extremely variable, dependent on the intrinsic effect of sedimentological and hydrological features on the local population dynamics. In the Clyde Sea for example, populations in the northern area have been found to be lower in density than the southern populations, but on average to be of a larger individual size (Tuck *et al*., 1997). This corresponded with significantly higher discards in abundance of *Nephrops* in the southern Clyde Sea, due to the higher numbers of small undersized individuals being caught (Bergmann *et al*., 2002). In a recent EC study, Wileman *et al.* (1999) found that the mean survival rate for discarded *Nephrops* was 31%. This is actually higher than that used by the ICES *Nephrops* stock assessment group who add a discard mortality of 75% to the fisheries mortality used. Wileman *et al.* (1999) suggest that their study may have underestimated discard mortality, because experimental tows were shorter than commercial tows, thus reducing the time spent on the deck. Another important finding of the study was that the survival rates of discarded *Nephrops* were significantly lower for females. The implications of this finding should be considered when modelling the population level response to fisheries mortality of *Nephrops* (See Section 5.2).

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

3.2 Mortality of non-target stocks

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

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

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

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

3.3 Mortality of benthic invertebrates in the tow path of the gear

A significant fraction of the benthic invertebrates that suffer direct mortality due to fishing are killed as a result of contact with the fishing gear as it passes over the seafloor. This is a much more important source of fisheries mortality to invertebrates than it is to demersal fish due to the largely sessile nature of benthic invertebrates. This 'unobserved mortality' is difficult to quantify and it is only in recent years that real progress has been made in bringing together the results of a number of disparate studies (Collie *et al*., 2000; Johnson, 2002; unpublished data Cost-Impact, EC Project number: Q5RS-2001-00993).

The only methods that can really be employed to quantify the absolute mortality in the towpath are through counts made by divers, or from remote video or submersibles following the passage of the gear (e.g. Caddy, 1973; Eleftheriou & Robertson, 1992; Hall-Spencer *et al*., 1999). Even then it is often difficult to establish whether an animal is actually dead or just damaged. Clearly if an animal is badly damaged it is likely that it will be vulnerable to predation or disease as a result of its injuries and thus will face secondary mortality as a consequence of fishing (Hill *et al*., 1996). However any subsequent predation mortality is an indirect effect of the fisheries disturbance and should not be counted in the quantification of the actual fisheries disturbance index (See Section 5). Where observed mortality cannot be quantified immediately, studies have calculated the percentage change in abundance, biomass or density of individual populations or communities, either before and after a fishing event or at fished and unfished (control) sites (See Lindeboom & de Groot, 1998 and references in Collie *et al.*, 2000).

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

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

There is also evidence that the mortality of benthic fauna in the path of the trawl is strongly size dependent (Engel & Kvitek, 1998; Kaiser *et al*., 2000; Bergman & van Santbrink, 2000; Duplisea *et al*., 2002). It is suggested that, within and among species, the mortality rates suffered by the smallest individuals may be lower because they may be pushed aside by the pressure wave in front of the trawl (Gilkinson *et al.*, 1998). This is only relevant, however, to the organisms that are epifaunal in habit or that live close to the surface of the sediment. For animals that are truly infaunal in habit, mortality of smaller individuals may actually be higher if there is a relationship between individual size and depth distribution. For example, Bergman & Hup (1992) found that the fishing mortality of small *Echinocardium cordatum* (Heart urchins) and *Lanice conchilega* (Sand mason worms) was much higher than the mortality in larger individuals. In studying the relationship between depth distribution and size of heart urchins, smaller individuals were found to have a mean depth distribution of 2-4cm, whilst the larger individuals had a mean depth distribution of 10-12cm, below the penetration level of the gear. It is clear that there will be difficulty experienced in trying to estimate the mortality sustained by each invertebrate population in the towpath of the gear due to the effect of the combination of the various characteristics that influence vulnerability. This is further complicated when the variation in actions of fishing gears and the influence of substrate type are taken into account (See Section 4.5).

Given the number of variables that appear to affect the population level mortality, the only viable way of determining towpath mortality is to bring together the results of all the disparate studies and to then analyse these to try to pick out consistent patterns. In recent years a number of individuals and projects have undertaken this task. In some cases studies are reviewed and conclusions drawn purely on qualitative evaluations of the combined results (e.g. Watling & Norse, 1998; Johnson, 2002; Thrush & Dayton, 2002). Collie *et al.* (2000) have however quantitatively analysed the combined findings of these studies using meta-analysis techniques and their work is now being updated and developed by the EC Cost-Impact project (EC Project: Q5RS-2001-00993). It is hoped that this work will lead to the potential to predict mortality of invertebrates in the towpath at a population level, given a particular fishing regime, with particular gears, in a specific habitat type. It is important to recognise, however, that only some of the studies included in these meta-analyses actually give absolute fishing mortality values. As described earlier, in many cases there is a delay between the fishing event and the measured mortality in the population, allowing for the incorporation of the community level response (See Section 5.2). If this significant source of fishing mortality is to be included in the determination of fisheries disturbance to benthic invertebrates it may necessary to make a number of assumptions about the level of mortality actually attributable to the fishing event. It is also very unlikely that it will ever be possible to make these sorts of predictions for all species that make up the diverse benthic communities of the North Sea. However, it is hoped that the science will develop towards the ability to predict mortality for characteristic species and associated functional groups if a particular gear is used in a particular habitat. This will enable a more ecologically meaningful inclusion of towpath mortality in the estimation of fisheries disturbance indices for benthic invertebrate communities.

As described in Section 2.3 for fish, there is also another element of unaccounted mortality for benthic invertebrates in the towpath of the gear. This is for those animals that actually pass through the fishing gear but then die merely as a result of the injuries they sustain in this process. As with fish, the experimental work that has been undertaken to try to quantify the proportion of escapees that die, has been restricted to commercially important species, in most cases one species, *Nephrops norvegicus*. Wileman *et al.* (1999) investigated the escapee mortality rate of *Nephrops*, with the assumption that survival rates may be lower than had been found for roundfish fisheries (see Section 2.3) due to the high quantities of abrasive material usually found in the codend of *Nephrops* trawls. This material includes shells, stones and various crustacea that are mixed in with the target species. Survival rates of *Nephrops* were however found to be comparable with roundfish, with a mean survival rate of 82%. In comparison with the mortality of discarded animals (mean 31% survival rate), the additional mortality associated with escapees from the gear was found to be insignificant from a stock assessment perspective. It is also suggested that the escapee mortality recorded in this experiment may be overestimated as the escapees were held in the codend cover for the duration of the trawl (2 hours). In the same study it had been found that there was a positive relationship between time spent in the cover and increased escapee mortality for roundfish (See Section 2.3).

In considering the actual ecological disturbance associated with particular gears, it will be important to consider whether this level of additional mortality makes a significant contribution. Certainly in considering the community level response following the fisheries disturbance (Section 5), it is likely that the escapees will be more vulnerable to predation. In the experiments conducted by Wileman *et al*. (1999) on *Nephrops* escapees, it was found that the tail flip mechanism (the escape response exhibited by *Nephrops*) was reduced by 53% for the first 2 hours following escape from the codend.

4. PREDICTING THE ECOLOGICAL DISTURBANCE OF FISHING USING FISHING EFFORT STATISTICS

4.1 Mapping fishing effort on a North Sea scale

Greenstreet & Rogers (2000) stated that fishing effort has never been evenly distributed across the North Sea. Different gears, directed at different target species, with differing levels of impact on the components of the ecosystem, have been used at varying intensities across the North Sea. In order to develop spatially and temporally resolved indices of the ecological disturbance of fishing on benthic fish and invertebrate communities, at the very least there is a clear need to obtain data for the amount of fishing effort in a given area at a given time. As described in Sections 4.4 and 4.5 below, more resolved information on the types of gear and the power of the vessels used would further improve the potential for developing ecologically meaningful indices. Most of the countries that fish within the North Sea record routine measures of fishing effort at the scale of the ICES rectangle. These data are however variable in the procedures and measures used to record the data and the length of time for which they are available (Greenstreet *et al*., 1999).

The longest time series available is for the effort of UK vessels landing in Scotland and Greenstreet *et al.* (1999) analysed trends in both the demersal and pelagic fleets over the period 1960-1994. This work has recently been updated to include the years 1995-1998 (Greenstreet *et al*., 2004). Jennings *et al*. (1999b) analysed international trends in demersal trawling over the shorter period 1977-1995. This analysis included effort data from English, German, Norwegian, Scottish and Welsh vessels over the entire period and also Danish and Dutch vessels between 1990-1995. No data were available from the Belgian and French fleets and it was felt that effort had been underestimated, potentially by >50% in the Southern North Sea area, due to the lack of these data. This international database has since been updated to include the year 1998 and the spatial distribution of effort in the North Sea for this period is given in Callaway *et al*. (2002).

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

4.2 Improving the accuracy of predicting effort distribution

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

These findings have real implications for the derivation of meaningful ecological indices of fisheries disturbance at the scale of the ICES rectangle. If, for example, a particular level of disturbance (predicted from the mean annual effort in hours fishing of that rectangle) is distributed evenly across the rectangle, areas that may in reality only be subject to very low effort may be overestimated, whilst areas of aggregated high effort may be underestimated. Rijnsdorp *et al.* (1998) cite a study by Rauck (1985) that predicted the level of beam trawling effort at the scale of the ICES rectangle. This study suggested that every square metre of the seabed was on average trawled 5-7 times per year. However, Rijnsdorp *et al.'*s (1998) model of effort distribution, based on the microscale data for the beamtrawl fleet, predicted that on average, only 1% of the area within a heavily fished rectangle was trawled more than 5 times a year. Given that it is thought that the percentage mortality sustained by an invertebrate population in the towpath of the gear is likely to depend on the frequency of trawling (See Section 3.3), estimating a reasonable level of mortality from that source is really dependent on a realistic distribution of effort. A number of studies have now used higher resolution effort distribution data to evaluate the disturbance of fishing in benthic invertebrate populations of the southern North Sea (Bergman & van Santbrink, 2000; Piet *et al*., 2000). It is clear from the results of these studies that the inclusion of high-resolution effort data significantly effects the estimation of the levels of mortality experienced by populations and communities at the scale of the ICES rectangle.

A number of different sources of data are now becoming available to track the microscale distribution of individual fleets. A proportion of the Dutch fleet has been tracked for over 10 years. Initially 'black boxes' (automated position recording systems) were installed on 10% of the fleet and these gave positions every 6 minutes to an accuracy of approximately 100m (data from 1993-2000). Since 2000 however, the microscale distribution of approximately 30% of the fleet has been available through a private agreement on access to VMS (European Community Satellite Vessel Monitoring System) data (G. Piet, *pers. comm.*). Since the 1st January 2000 it has been compulsory for EC registered fishing vessels over 24m to report their location every 2 hours, using the VMS system. Exceptions include vessels that undertake trips with a duration under 24 hours, or that fish exclusively within territorial waters. Due to problems instigating the system on an international scale, reliable data are only available from July 2000 (Dinmore *et al*., 2003).

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

4.3 Changes in fishing practices over time

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

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

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

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

4.4 Predicting demersal fish mortality from fishing effort statistics

The determination of the level of fishing disturbance to demersal fish communities in a given area will depend on the ability to accurately predict mortality from the fleets operating in that area. In reviewing the disturbance of demersal fish by fishing, it is clear that there are a number of sources of mortality that must be accounted for. These are the mortality of all fish, both target and non-target that are landed, the mortality of discarded bycatch, also including target and non-target species, and the mortality of fish that escape from the gear but subsequently die (See Section 2).

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

In order to determine levels of mortality of demersal fish discarded by the fleets fishing in a given area, both the levels of discards per species and the survivorship of those species should be known. For recent years, the numbers of each fish species discarded per trip, should be recorded in the discard monitoring schemes of each country, with a registered fishing fleet in the EC. However, these schemes only cover a sample of all trips carried out by a fleet in a year and so mortality of each species would have to be raised to the scale of the fleet. In order to predict the level of mortality to the demersal fish community sustained in the discards, it will be important to consider the influence of a number of factors on the relative abundance of different species making up the bycatch. For marketable species, changes in quota and market value are likely to affect the level of discarding, whilst the habitat type of the area fished and the targeted stock of a fleet is thought to affect levels of overall discard mortality. Although numbers of discards do not represent absolute mortality of those animals, it appears that there are only a limited number of studies of the survivorship of species following discarding. The results of these studies will be considered and the availability of any further data on survivorship of discarded fish investigated. Information on the mortality sustained by escapees seems to be equally sparse but initial results do suggest that survivorship of this element will be much higher than that of the discarded catch (See Section 2).

4.5 Predicting benthic invertebrate mortality from fishing effort statistics

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

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

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

The spatial distribution of benthic invertebrates is known to be patchy and it is now thought that the distribution of effort can also be highly aggregated and heterogeneous (See Section 4.2). It will therefore also be particularly important, where possible, to resolve microscale distribution of fishing effort for the determination of benthic invertebrate disturbance indices. The resolution of data in order to account for the overlap in different communities with different levels of fishing effort will help to improve the precision of disturbance indices (Rijnsdorp *et al*., 1998; Piet *et al*., 2000). In trying to do this, the availability of data on the spatial distribution of particular community types will also be vitally important. Having microscale distribution of the effort data will provide little help if it is not known how communities are distributed within that area. A number of key papers that describe the distribution of characteristic infaunal and epifaunal invertebrate communities are available at the scale of the North Sea (Duineveld *et al*., 1991; Künitzer *et al.,* 1992; Callaway *et al*., 2002). It will be important to consult all known sources of data on the distribution of both epifaunal and invertebrate communities before determining the disturbance indices.

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

5. RESPONSE OF DEMERSAL FISH AND BENTHIC INVERTEBRATE COMMUNITIES TO FISHERIES DISTURBANCE

5.1 Implications of fisheries disturbance to demersal fish communities

It is certain that the community composition of demersal fish species in the North Sea has changed in the last 20-30 years (Greenstreet & Hall, 1996; Heessen & Daan, 1996; Greenstreet *et al.*, 1999; Greenstreet & Rogers, 2000; Clark & Frid, 2001). Some of the evidence for why these changes have occurred infers a role for the effects of fisheries disturbance. This may be through the alteration in competition that has resulted from the removal of large numbers of targeted species or from changes in the availability of food resources due to the fisheries disturbance of benthic invertebrate communities. Jennings *et al.* (1998) examined the differential effects of fishing on individual species with contrasting life histories. This work suggested that those species that decreased in abundance relative to their nearest relative, matured later at a greater size, grew more slowly towards a greater maximum size and had lower rates of potential population increase. It was proposed that trends in community structure could be predicted from the differential responses of related species to fishing. Results agree with the prediction that fishing has greater effects on slow growing, larger species with later maturity and lower rates of potential population increase (Jennings *et al.*, 1999a). More recent work concurs with this idea and suggests that the differential effects of fishing on species and populations with different life histories is a stronger and more universal indicator of fishing effects than changes in the mean trophic level (Jennings *et al.*, 2002a).

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

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

There is far more information, however, on the response of scavenging and predatory demersal fish species to the increase in food resources left in the track of the fishing gear. As has been described in several of the earlier sections, considerable numbers of dead or dying fish and invertebrates are left in the wake of fishing vessels. These are the animals either discarded from the vessel or those that are killed or injured in the towpath of the gear. Groenewold & Fonds (2000) calculated that over 10% of the total annual secondary production of macrobenthic invertebrates becomes available, as damaged or displaced animals in the passage of a single beam trawl in the southern North Sea. There is clear evidence that mobile scavengers, including some demersal fish species, will actively move into a trawled area to take advantage of this increase in resources (Kaiser & Spencer, 1996a; Fonds & Groenewold, 2000). Highly mobile predators such as fish have been recorded to arrive at the fished area within 30 minutes of the disturbance occurring (Kaiser & Spencer, 1996a). A number of studies have investigated the ability of invertebrates to perform classic escape responses following either discarding from a vessel, or contact in the towpath of the gear. In most cases these studies have investigated target species and in all cases the escape response is greatly reduced, even when physical damage to the organisms is low, suggesting that specimens will be vulnerable to greater levels of predation (Ramsay & Kaiser, 1998; Coffen-Smout & Rees, 1999; Wileman *et al*., 1999; Jenkins & Brand, 2001). For some species damage to individuals is so severe as to prohibit any escape response in over a quarter of all specimens in the track of the gear (Jenkins *et al.*, 2001). The short term increase in food resources for scavenging and predatory demersal fish are likely to be significant but it will be important to try to quantify the relative importance at the community level over a wider area.

Even more difficult to quantify is the effect of the overall change in benthic invertebrate community, as a response to fisheries mortality, on the demersal fish community (See Section 5.2). Increased growth rates in some flatfishes have been linked to improved feeding conditions

(Rijnsdorp & Vingerhoed, 2001; Rijnsdorp & Van Leeuwen, 1996) that are thought to be as a result of increases in the abundances of small polychaetes, over the same time period (Rijnsdorp & Van Leeuwen, 1996). These increases in the relative abundance of fast growing polychaetes in the benthic invertebrate community have been linked to sustained fisheries disturbance, but as pointed out by Jennings *et al*. (2002b), the analyses are complicated by increases in primary production over the same period.

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

5.2 Implications of fisheries disturbance to benthic invertebrate communities

Fisheries mortality of benthic invertebrates is largely an unknown quantity at the scale of the North Sea. However, it is clear that for some combinations of fishing gear and habitat type, mortality both in the gear and in the towpath of the gear is likely to be high for some components of the community (See Sections 3 and 4.5). The overall benthic invertebrate community level response to fisheries disturbance will depend on a number of factors. These include the absolute mortality following the passage of the gear, the effect of fisheries mortality to the demersal fish community and the effects of the alteration of habitat type following the passage of the gear. It is likely that the community level response will also vary dependent on the influence of a number of other drivers that structure the community. These include the level of local productivity, the local availability of propagules for immigration into the disturbed area and the influence of hydrography and climate.

In many areas of the North Sea, time series studies have inferred a role for fisheries in the long-term changes in community composition seen. This shift in composition could be a result of sustained fishing disturbance but the influence of climate and other anthropogenic drivers such as pollution and eutrophication cannot be discounted (Engel & Kvitek, 1998; Kaiser *et al*., 2000; Bergman & van Santbrink, 2000; Kröncke & Bergfeld, 2001). There is a suggestion that benthic invertebrate communities have changed from those dominated by low productive, slowly reproducing organisms to quickly reproducing, opportunistic species. It is likely that the larger, slow-growing species (k-strategists) will be particularly vulnerable to sustained levels of mortality, whilst smaller individuals and species can endure higher mortality rates (Gilkensen *et al.*, 1998). Clearly though, fishing may not be the only factor increasing mortality in these communities and the development of fisheries disturbance indices will help to elucidate how significant the mortality resulting form fishing is. Some studies have hypothesised that many of the large, high biomass infauna burrow below the depth that most bottom fishing gears will penetrate, confounding the suggestion that changes in size structure of benthic invertebrate communities may be a response to fisheries disturbance. Hall-Spencer *et al*. (2001) do however point out that although the large adults of some species do pass below the gear, the populations may still decline because there is reduced recruitment of juveniles that do live within the penetration depth of the gear. Jennings *et al*. (2001) suggest that the differential vulnerability of species to trawling leads to lower biomass and production of communities in heavily trawled areas and a dominance by smaller, faster growing individuals and species. Dinmore *et al.* (2003) describe Duplisea *et al.* 's (2002) size-based model that was used to assess the impacts of trawling on benthic production. For invertebrates in the range of 1μg to 80g (shell free wet weight), the model predicted that larger species could only survive in some fishing grounds because trawling disturbance was patchy.

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

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

There have been a number of studies on the response of benthic invertebrate scavengers to the availability of moribund material in the towpath of the gear and the deposition of discards on the seafloor following the release of discarded bycatch over the side of the vessel. Some of this material will float and large quantities of discarded fish are taken by scavenging seabirds (Hudson & Furness, 1988; Garthe *et al.*, 1996). The remaining discards, except for a small amount taken by fish and marine mammals in the water column, will however fall to the seafloor. Although some discards will survive, many will be dead already or will have suffered high levels of physical stress and thus will be vulnerable to predation from demersal scavengers. As described in Section 5.1, a study of the moribund invertebrate material left in the trawl track of a southern North Sea beam trawl fishery was comparable to greater than 10% of secondary macrofaunal production available in the area (Groenewold & Fonds, 2000).

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

Although this review does not implicitly cover the alteration of habitat that results from demersal fishing, there is evidence in the literature to suggest that there will be an overall community level response of benthic invertebrates to this kind of fisheries disturbance (For reviews see, Watling & Norse, 1998; Johnson, 2002; Thrush & Dayton, 2002). The physical alterations caused by the passing of the gear will in most cases change heterogeneity of the sediment surface, alter the texture (particle size composition) of the sediments and change the structure available to biota as habitat. Since the distribution of most benthic macrofaunal species in the North Sea is related directly to sediment particle-size composition and organic content (Duineveld *et al*., 1991), the physical disturbance associated with fishing effects will inevitably have consequences on the structure of the benthic invertebrate community. Although alteration of habitat may affect particular life stages of demersal fish (See Section 5.1 above), benthic invertebrates have close associations with the benthic habitat throughout their lifecycle. Thus, it is perhaps even more important that the implications of habitat change are included in the fisheries disturbance indices derived for benthic invertebrates.

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