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Report

Number: C087/05

A method to quantify the trawl fisheries induced mortality of benthos and fish

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Commissioned by: RIKZ
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Project number: 3251212122

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Date: 17 december 2005

Number of copies:	32
Number of pages:	58
Number of tables:	5
Number of figures:	24
Number of annexes:	2

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Summary

Effects of bottom trawling on benthic life have been reviewed in several recent papers. However, this information is usually not presented in a manner that makes it applicable in a management context. In this report we describe what type of information is needed and how this can be used as part of an ecosystem approach in fisheries management.

Effects of fishing gear on marine communities can be separated in direct and indirect effects. Direct effects are the removal, damaging or injury of benthic organisms and fish, as well as destruction or modification of the habitat. Indirect effects are the secondary effects e.g. through predator-prey relationships or loss of physical structure. In this report we are focussing only on the direct effects of fishing, expressed as the population mortality of benthos and fish species. Depending on their life history characteristics different populations may withstand different levels of additional fisheries-induced mortalities. How these levels are determined by the life-history characteristics are considered outside the scope of this report.

This report describes an approach that provides a spatially explicit quantitative estimate of the impact of bottom trawling on both benthos and fish, that requires information on:

- The absolute abundance per species per spatial unit (ICES rectangle). This can be based on survey data in combination with information on the catchability of the surveys.
- The frequency with which the spatial unit is trawled. An overview of recent information on the intensity of fishing and its spatial distribution in the North Sea is carried out using data from the Vessel Monitoring System (VMS), a data collection system through satellite. Fishing intensity is presented on a quarterly basis for the period 2000-2004.
- The impact of the singular passing of a gear: expressed as direct mortalities of benthos and fish caused by bottom trawling. This was based on a review of the direct mortalities of benthos based on a meta-database compiled by Mike Kaiser and Hilmar Hinze consisting of 55 references with a total of 2474 datapoints. After selection according to relevant criteria a set of 16 references and 316 datapoints remained. Mortalities turned out to be highly variable and habitat and gear related. In general mortalities for beam trawls are higher than for otter trawls. For fish there are no estimates of the encounter mortality caused by the passing of the gear. Instead we use catchability, where mortality of fish in a fishing gear is determined by factors such as: positioning in the water column, herding, escape below footrope, retention in the net. In addition we also present a summary of studies that assessed the effects on fish that have escaped from a bottom trawl.

In this approach the total mortality of a species at the scale of the North Sea is the sum of the mortalities over all spatial units.

This approach is illustrated in a study that estimates mortalities of non-target fish species (Piet et al. MS-b). Results from this study are still preliminary but indicate that this approach can deliver realistic estimates of fishing mortality. In summary they show that according to this model the fishing mortality increases with increasing size of the fish, first for the flatfish and rays followed by the roundfish and sharks. Mortality of the larger size-groups is highest for the

fish at about 69% for flatfish to 75% for roundfish and lower for the elasmobranchs at about 64% for the sharks, 43% for the rays and 10% for pelagics.

Following from this approach we present a framework that distinguishes six levels of increasingly better fishing pressure indicators with increasingly higher levels of information content. These levels are (in order of increasing amount of information necessary): fleet capacity, days-at-sea, the proportion of time actually spent fishing, the frequency with which an area is fished, the spatial distribution of effort, fishing-induced mortality and finally at the highest level the spatial distribution of the ecosystem component to be studied relative to that of the fishery.

1. General introduction

RIKZ commissioned a literature study to synthesize existing knowledge on the effects of bottom disturbance by fishing on benthic life in the North Sea. Such a study has just been carried out by English researchers that had submitted a manuscript entitled: Global analysis of the response and recovery of benthic biota to fishing (Kaiser et al. MS). This manuscript provides a meta-analysis of 55 studies on the effects of fishing on benthos. In total a series of 2474 datapoints were brought together in a database. The paper focuses on the interactions between different factors such as fishing gear and habitat and therefore provides a tool for policy makers to understand why fishing activities are most deleterious in certain categories of habitat. In addition to this study there are several other recent reviews (Dayton et al. 1995b, Hall 1999). Jennings and Kaiser (1998a) have summarized effects of fishing of all types of fishing (including bottom trawling) on marine ecosystems.

However, this information is usually not presented in a manner that makes it applicable in a management context. In this report we present an overview of the method that has recently been developed to estimate the impact of fishing (i.e. bottom trawling) on both benthos and fish in terms of the proportion removed or mortality. Instead of describing this method in detail we present the conceptual framework of the approach and illustrate it with a case study.

A problem is that almost all impact studies have been conducted in areas that have already been fished over many years. This implies that most of the studies only indicate short term impacts on already influenced populations. They are not conclusive about long term impacts, neither at species or ecosystem level.

Many fishing impact studies are only applicable to a certain area/habitat and a few taxa. To be able to investigate the impact on population level it is necessary to combine spatial distributions of species and fishing impact with mortalities specified per species size and gear. We illustrate this conceptual framework by a case study on the impact of bottom trawling on fish. In the model (Piet et al. MS-b) species-specific mortalities are used as input.

To this end we were able to use the meta-database that was compiled for the review paper by Kaiser et al to give an overview of species-specific mortality rates for benthos species. In addition we describe a method to compile size- and species specific estimates for (mostly non-target) fish species.

In addition RIVO was also asked to give an overview of possible indicators for fishing. For information on indicators of fishing activity we used a study that was recently carried out to support an ecosystem approach to fisheries management (Piet et al. MS-a).

Effects of fishing gear on marine communities can be separated in direct effects and indirect effects. All towed bottom gears dig into the seabed surface to varying degrees. Apart from the removal, damaging or destroying of benthic organisms and fish, also the habitat structure is changed. Resuspension, transport and deposition of sediment that are caused by trawl nets may affect settlement and feeding of benthos and fish (refs in (Jennings and Kaiser 1998a).

Other indirect effects may work via the foodweb as other species with which a predator-prey relationship exists are affected by the fisheries. In this report we are not dealing with these indirect effects but focus on the direct effects of fishing.

To what extent a species can withstand this additional (either direct or indirect) fishing mortality is determined by its life-history characteristics (Jennings et al. 1999b). How these life-history characteristics determine which levels are sustainable is considered outside the scope of this report.

2. Conceptual framework to assess the impact of bottom trawling on benthos and fish

2.1 Introduction

Bottom perturbation caused by towed fishing gear is considered to be the most important source of disturbance to the seabed. In order to ensure sufficient catches of target species the gears are towed in direct contact with the seafloor. Apart from the removal of the target species, also non target species, both fish and benthos, are caught and discarded (direct effects). In addition the seabed is disturbed and structures are damaged, causing changes in habitats and ultimately benthic communities (indirect effects). Although the impact of trawling on non-target benthic species has received a lot of attention (Dayton et al. 1995b, Jennings and Kaiser 1998b, Hall 1999), this was much less the case for fishing-induced mortality of non-target fish species.

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

The extensive sets of data that exist for the target species can be used for (Multi-Species) Virtual Population Analysis, (MS)VPA, to estimate stock abundance and fishing mortality, do not exist for the non-target species. Yet these species not only make up a significant part of the biomass but also fulfil an important role in the functioning of the ecosystem and are often of interest from a conservation perspective (e.g. sensitive species like most elasmobranchs).

Within an ecosystem approach in fisheries management the effects of fishing on all ecosystem components (including non-target fish and benthos) will need to be considered. The approach presented here allows estimation the impact of fishing (i.e. bottom trawling) on some of the ecosystem components most likely to be directly affected by fishing..

2.2 Approach

For benthic species this approach has already been applied and typically involves a combination of data on the abundance of the biota, level of fishing effort and the impact of a unit of effort on these biota (Bergman and van Santbrink 2000a, Piet et al. 2000, Duplisea et al. 2002). A similar approach is developed for fish species where the direct effects of fishing are expressed as the mortality of a specific fish species. The approach incorporates a spatial component in that the North Sea is divided into spatial units (e.g. ICES rectangles) and for each unit the mortality of each of these species is determined by:

- The absolute abundance of that species; i.e. the number of fish present in an ICES rectangle. As this abundance is based on survey data which are stratified according to ICES rectangles, this is the resolution used in the model.
- The frequency with which the spatial unit is trawled; this is a measure for the amount of effort in an area.
- The impact of the singular passing of a gear. For fish species this is expressed as the fraction retained by the gear; this is the catch efficiency of a trawl gear. Catch efficiency ranges from 0 (no effect) to 1 (maximum effect). For benthos species this is expressed as the mortality (proportion of initial density in the trawl track)
- The total mortality of a species in the North Sea is the sum of the mortalities over all spatial units.

In contrast to benthos for which spatially disaggregated, absolute density estimates and quantitative estimates of the impact of a unit of effort (i.e. the single passing of a trawl) are available (Thrush et al. 1998; Collie et al. 2000; Kaiser et al. MS) this is not the case for non-target fish. Therefore we developed a methodology aimed at delivering this type of information. The result is a spatially disaggregated direct effects model (DEM) which is used to estimate the size- and species-specific mortality caused by bottom trawling.

The abundance of marine ecosystem components is often difficult to determine simply because they are below the surface of the water. For benthos it is relatively easy as they are more stationary than fish and several abundance estimates exist from sampling a specific area with corers, grabs and fine-meshed beam trawls (Duineveld et al. 1991, Bergman and van Santbrink 1998). For fish, however, these techniques can not be applied and abundance estimates are based on MSVPA and only exist for the main commercial species.

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

Fishing mortality estimates for the North Sea are available for 10 commercial fish species that are assessed routinely to provide annual advice on Total Allowable Catches (TACs) (ICES 2002, 2003). The principal method used is (Multi-Species) Virtual Population Analysis (MSVPA), which requires reliable estimates of the age composition of the total international catches and allows an evaluation of the historic development of fishing mortality and stock numbers by age group up to the present day. For non-target species, however, the data necessary to run VPA-type models are usually not available and other approaches are therefore required.

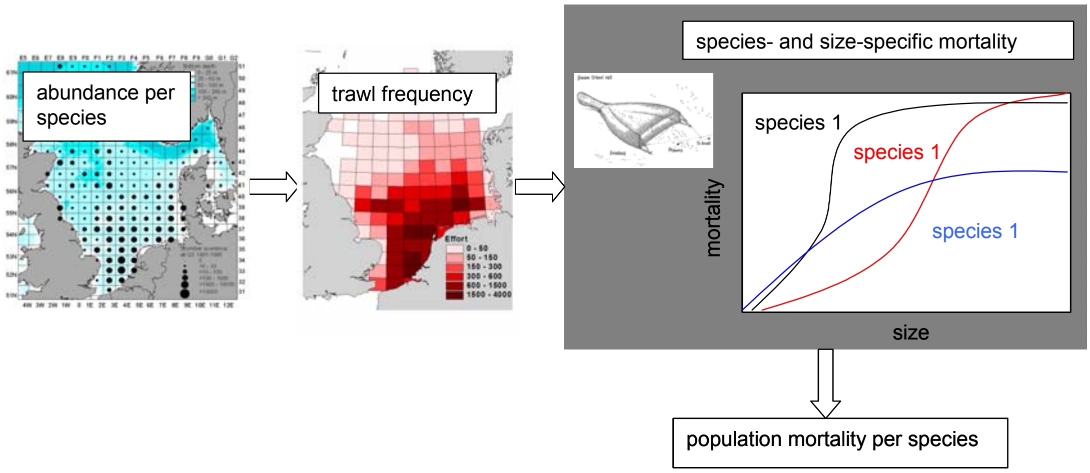


Figure 2.1. Figure to illustrate the approach to arrive at spatial disaggregated mortality of a species in the North Sea. Per spatial unit (ICES squares) the absolute abundance of every species, is combined with the frequency with which the spatial unit is trawled and the impact of the singular passing of a gear. The total mortality of a species in the North Sea is the sum of the mortalities over all spatial units. In combination with species-specific life history characteristics the longterm effect on populations can be assessed.

3. Absolute abundance

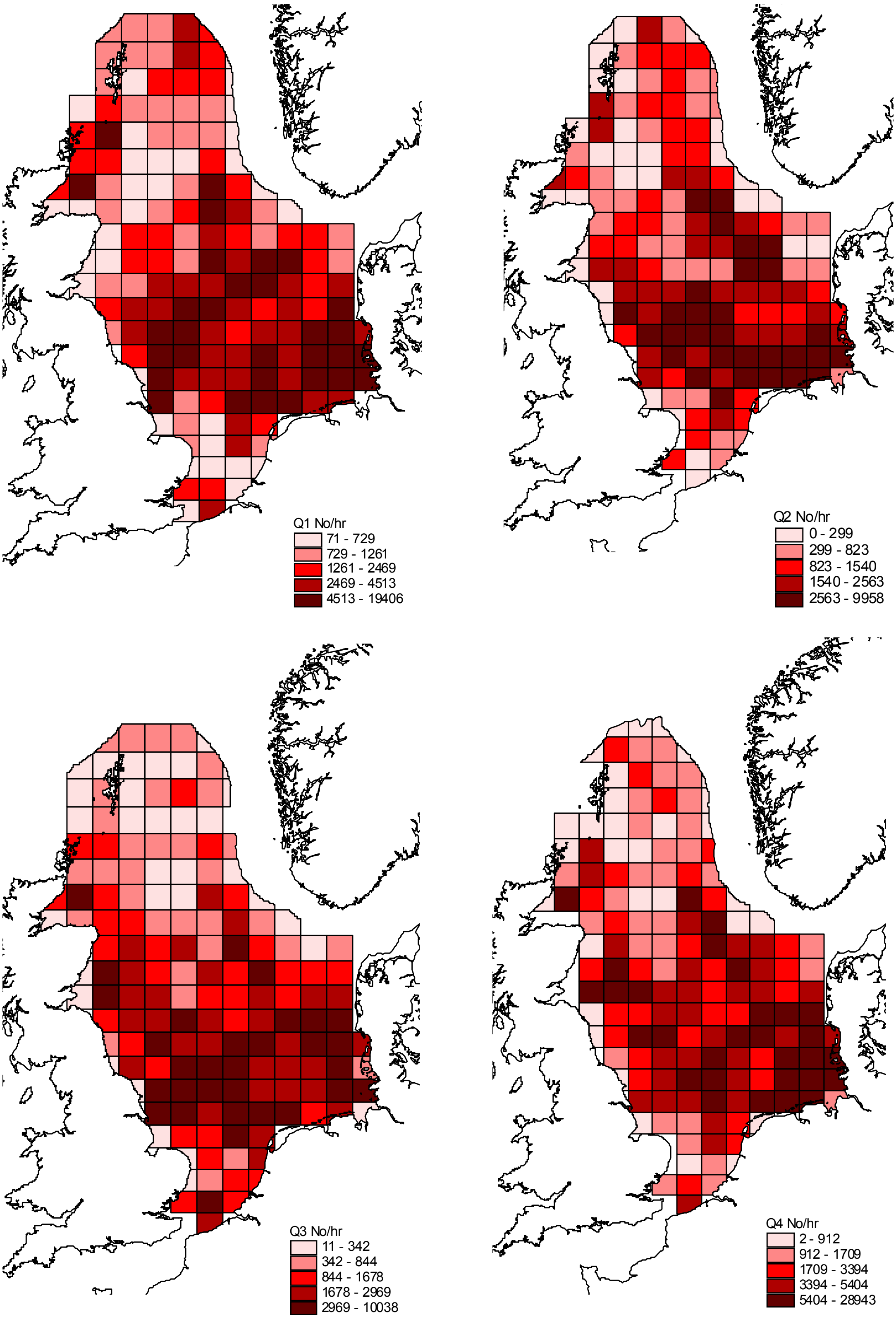
To estimate the abundance of non-target species we applied a modified version of the method developed by (Sparholt 1990). We followed Sparholt (1990) in that we estimated the abundance of non-target species by combining MSVPA-based abundance estimates of target species with survey catches that include both target and non-target species but improved the method in that we used different surveys for different groups of species and retained a size/age component that is inherent to the survey and MSVPA data. Sparholt (1990) distinguished the following groups: (1) cod , haddock, whiting , saithe; (2) norway pout; (3) herring, sprat; (4) sandeel; (5) mackerel; (6) plaice; (7) sole. VPA and MSVPA are assumed to provide the most accurate estimates of stock abundance at age in the North Sea of a suite of commercial species. Both approaches provide comparable abundance estimates but an advantage of MSVPA is that all abundance estimates are standardized to one area (i.e. ICES area IV) and on a quarterly basis. The catch rates of most commercial species-at-age in the North Sea are based on two surveys:

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

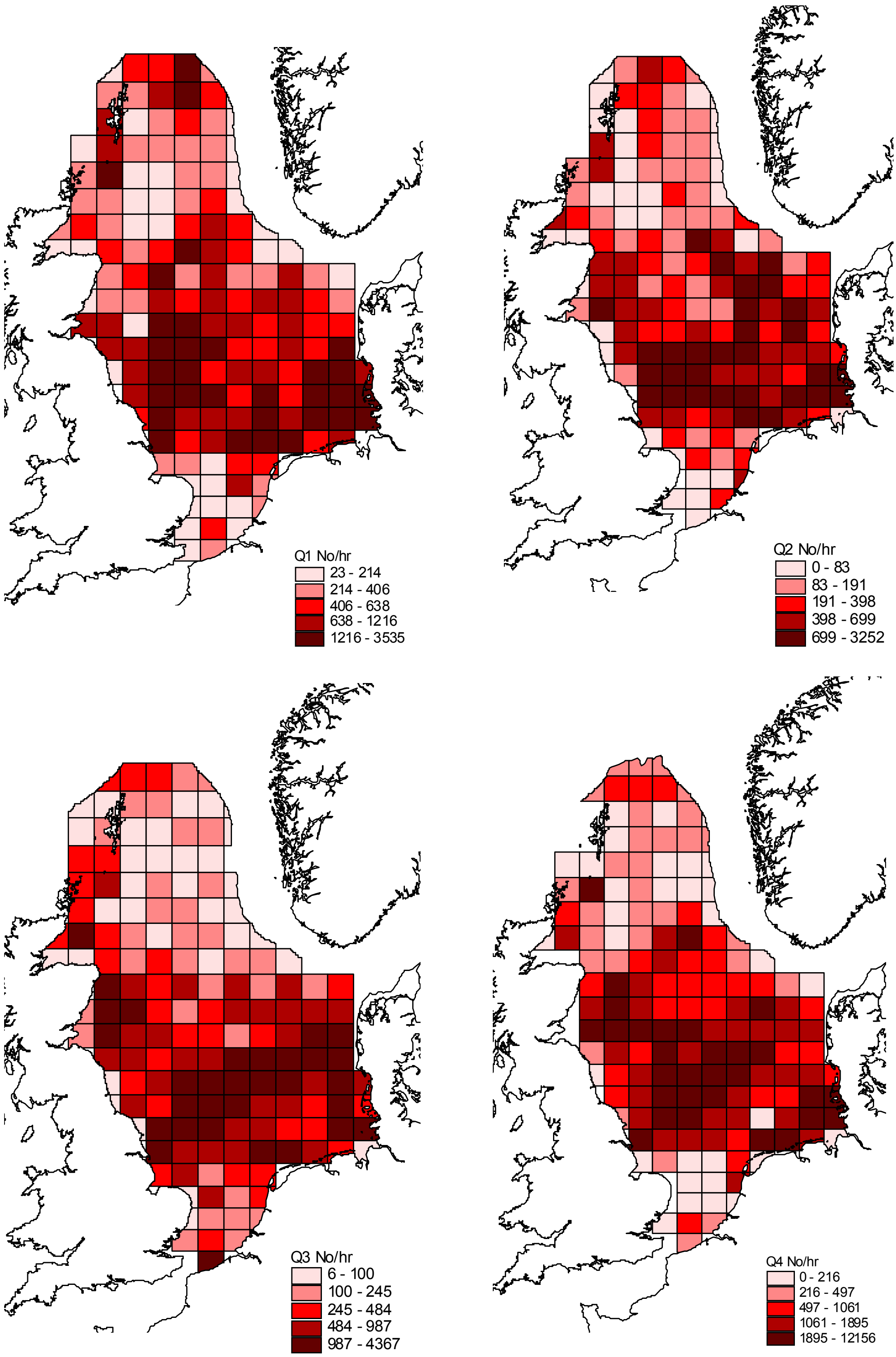
Because the MSVPA estimates of stock abundance are all fixed on the first day of the year the assumption is that this date is best represented by the 1st quarter IBTS catches. As the BTS takes place in the 3rd quarter the MSVPA abundance in the 3rd quarter was used. For the period 1991-1996 the IBTS was conducted on a quarterly basis. Results from that can be used to create quarterly distribution maps (see fig. 3.1).

Fig. 3.1. Quarterly distribution maps of all non-target demersal species in different size-groups according to the IBTS.

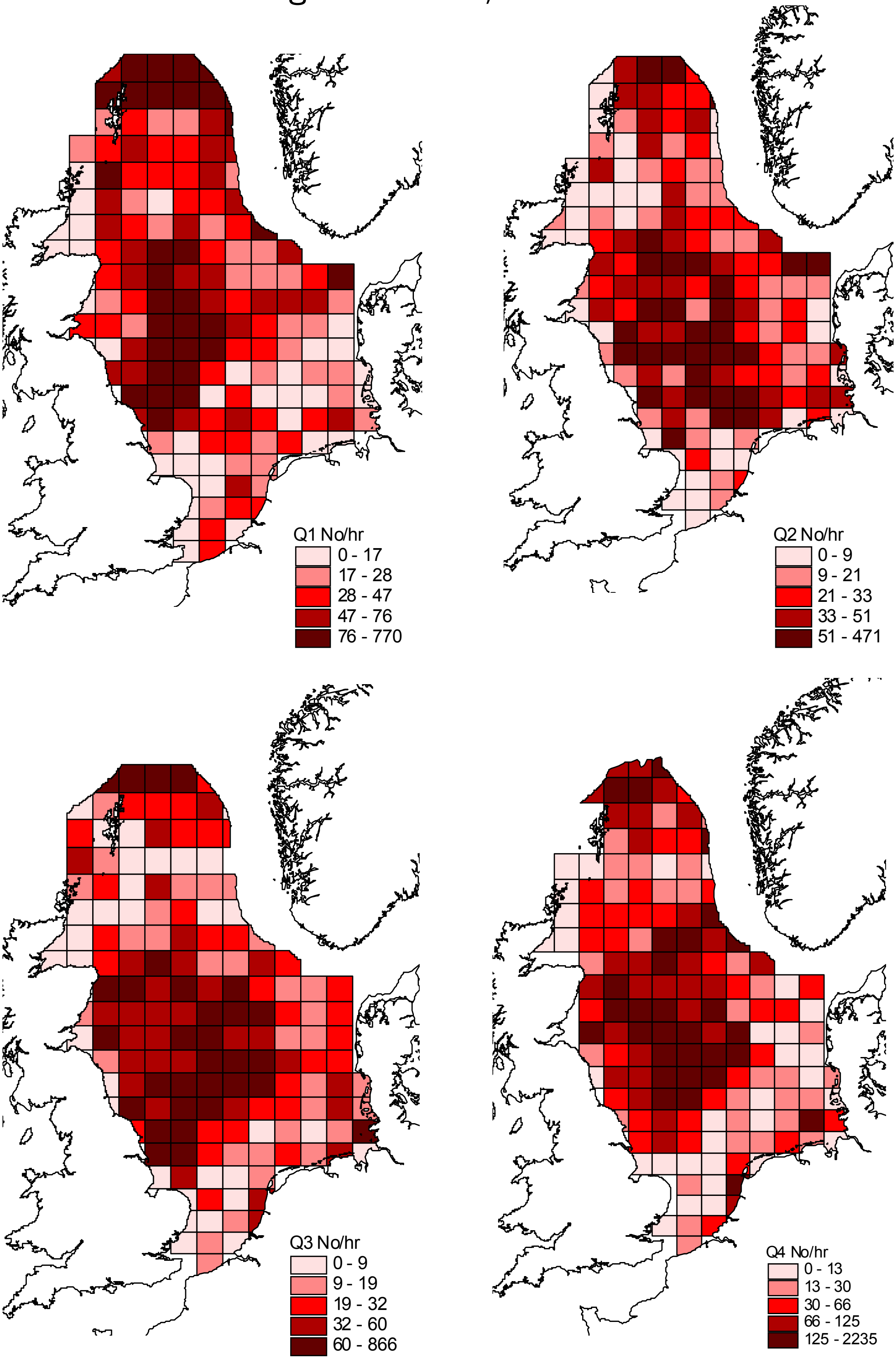
IBTS Non-target demersal, 10 – 20 cm.



IBTS Non-target demersal, 20 – 30 cm.



IBTS Non-target demersal, 30+ cm.



4. Fishing effort in 2000-2004

4.1 Methods

European inspection services monitor the spatial distribution of (fishing) vessels by means of a Vessel Monitoring System (VMS) through satellite. VMS data became available from 2000 onwards when positions of all EU vessels >24 m were recorded for enforcement purposes. Since September 2003 also positions of vessels >21 m were recorded. In this study we used data from the period 2000-2004. The data of Dutch fishing vessels are available for research purposes after getting permission from individual vessel owners.

The Dutch beam trawl fleet distinguishes two segments based on engine power: (1) euro cutters with engine-power below 300 Hp and (2) large cutters with higher engine-power.

The activity of vessels (i.e. fishing, steaming or not moving) is derived from their speed. Euro cutters are assumed to be fishing when their speed is between 3 and 6 nautical miles per hour; large cutters are assumed to be fishing when their speed is between 4 and 8 nautical miles per hour. However, registration of speed is not obliged, so not for all registrations the activity of a vessel is known. Moreover, by using speed as an estimator for the activity, it is possible that fishing effort is underestimated. If a vessel is fishing at a speed outside the fishing speed ranges we use, they are registered as steaming or not moving.

Note that the data comprise information from a sample of the fleet. At the moment data from 50% of the effort of euro cutters and 35-40% of the effort by large cutters is covered in the VMS dataset (both only using beam trawl, fig. 4.1). The sample is biased towards fishers from the southern harbours, i.e. vessels from Urk (that generally fish in the northern part of the North Sea) are underrepresented.

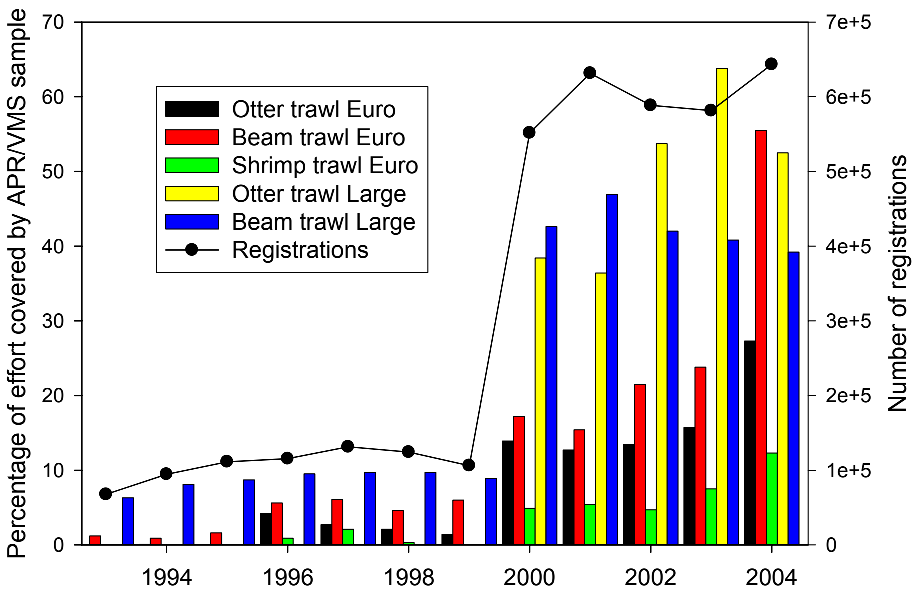


Fig. 4.1. Percentage of total effort that is covered by VMS.

4.2 Results

Fishing activity of vessels >300 Hp in winter (quarter 1) is concentrated in the southern part of the North Sea (figures 4.2-4.5). In spring vessels tend to move northwards and extend their fishing area following the movement of plaice. The ≤300 Hp vessels mainly fish along the Dutch and Danish coast and in the Plaice Box, with scattered activity in the central North Sea (figures 4.6-4.9).

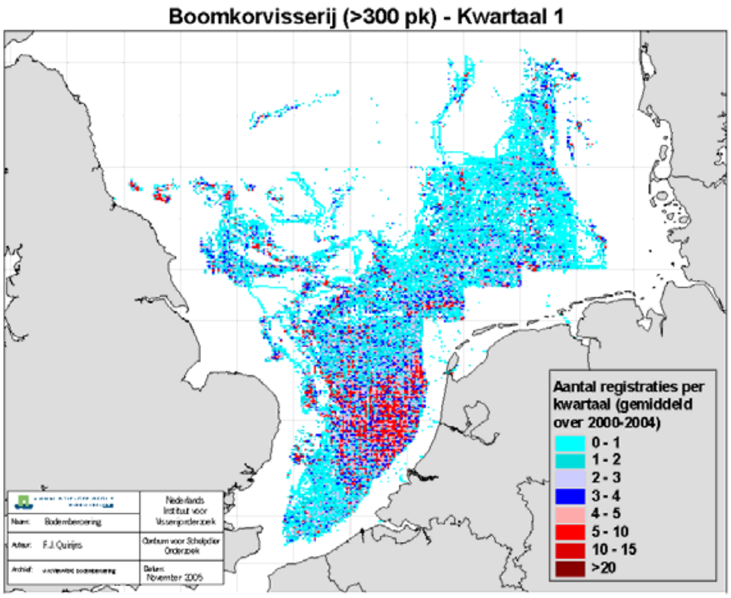


Figure 4.2 Number of registrations of vessels >300 HP per quarter in January-March (averaged over 2000-2004).

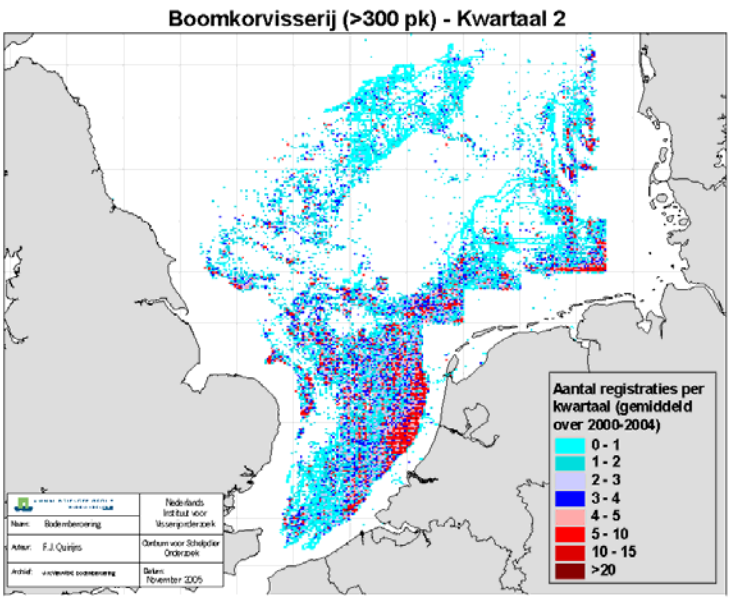


Figure 4.3. Number of registrations of vessels >300 HP per quarter in April-June (averaged over 2000-2004).

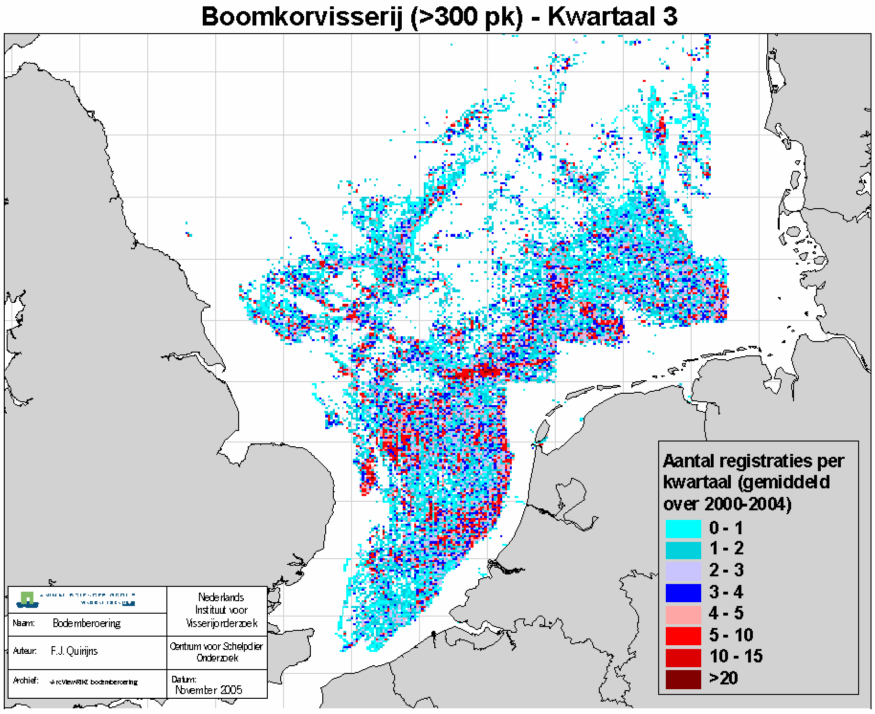


Figure 4.4. Number of registrations of vessels >300 HP per quarter in July-September (averaged over 2000-2004).

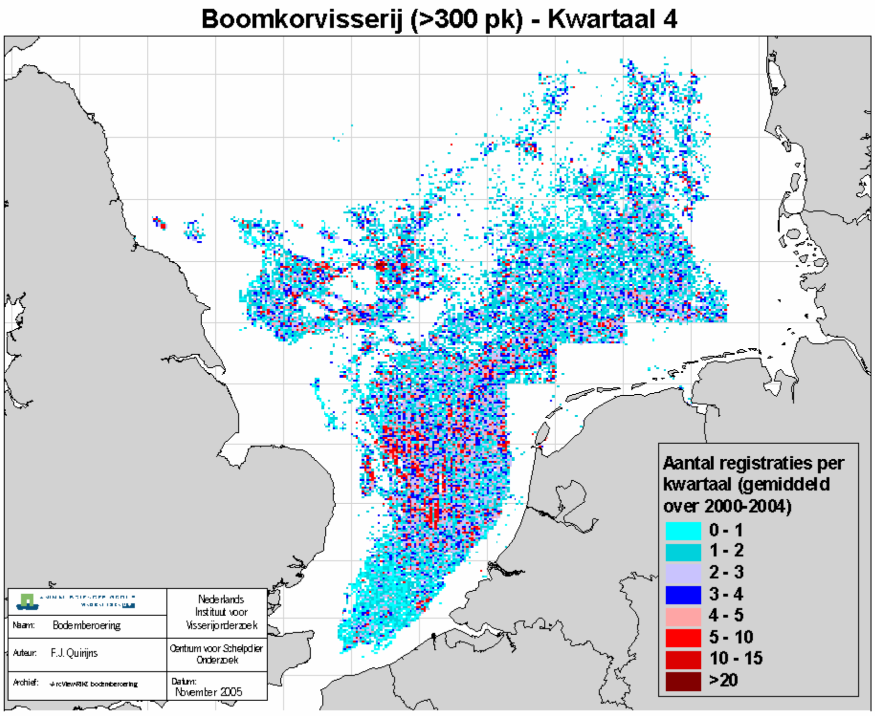


Figure 4.5. Number of registrations of vessels >300 HP per quarter in October-December (averaged over 2000-2004).

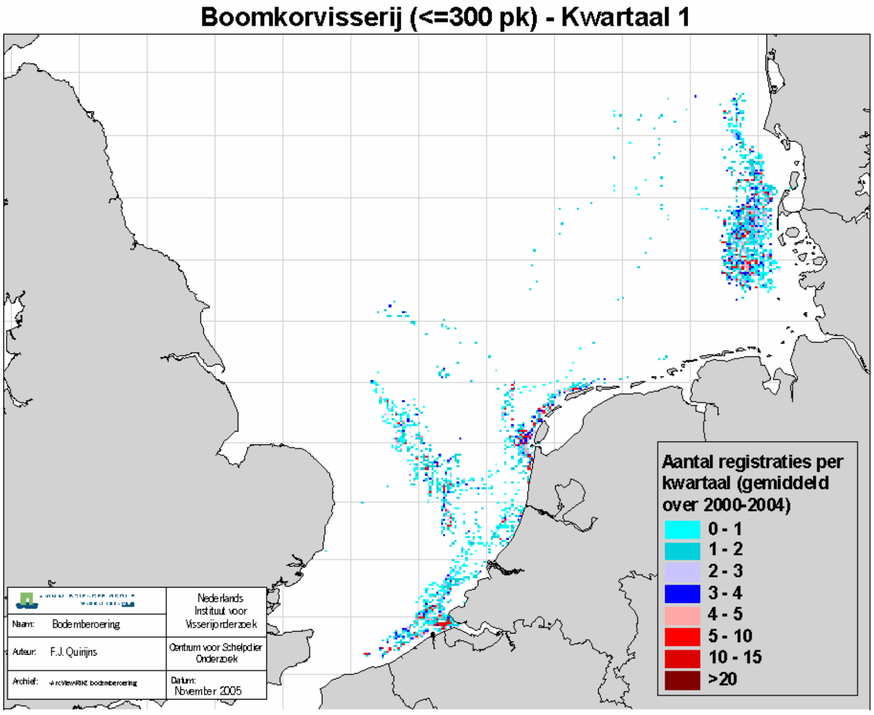


Figure 4.6. Number of registrations of vessels ≤300 HP per quarter in January-March (averaged over 2000-2004).

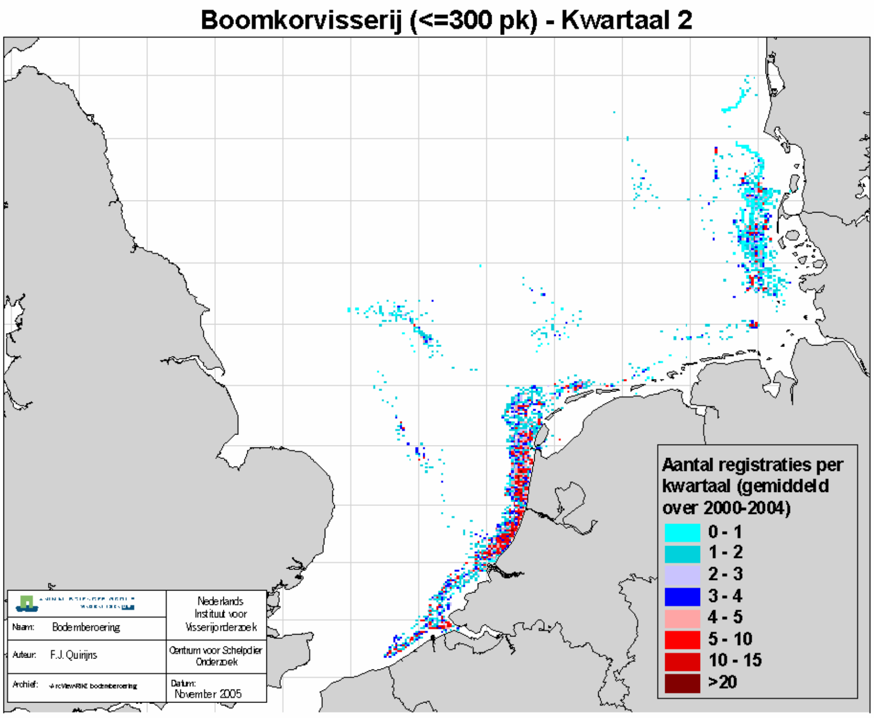


Figure 4.7. Number of registrations of vessels ≤300 HP per quarter in April-June (averaged over 2000-2004).

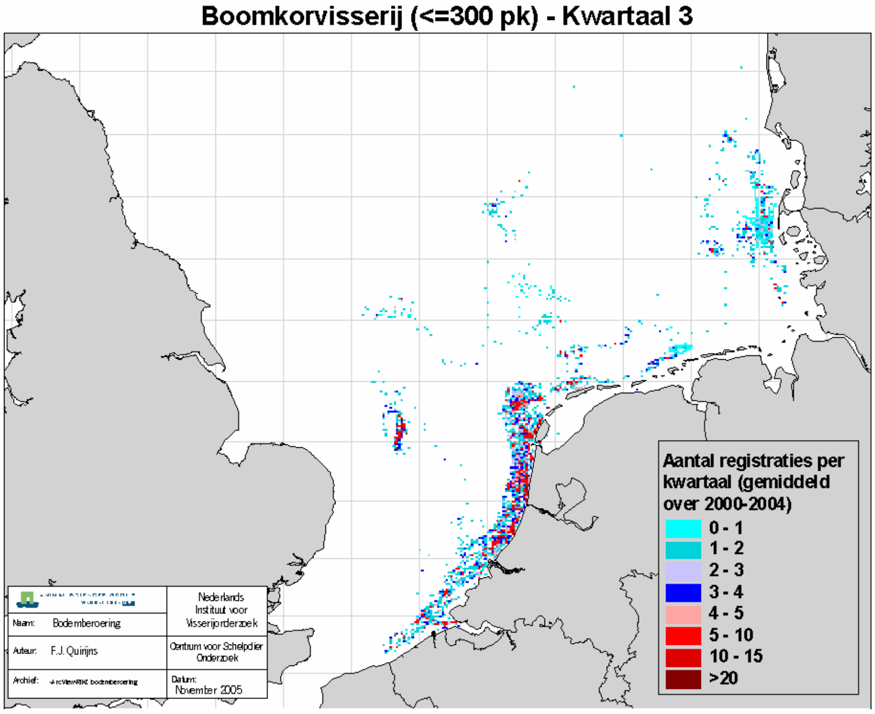


Figure 4.8. Number of registrations of vessels ≤300 HP per quarter in July-September (averaged over 2000-2004).

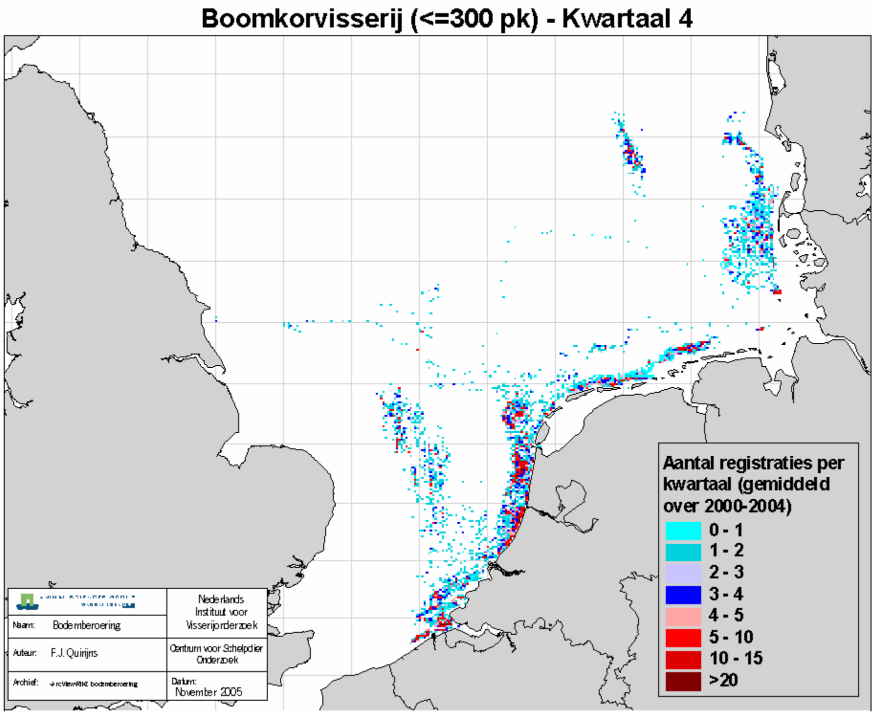


Figure 4.9. Number of registrations of vessels ≤300 HP per quarter in October-December (averaged over 2000-2004).

5. Impact of gear: size- and species specific estimates of mortality of benthos and fish

5.1 Direct mortality of benthos

5.1.1 Methods

The analysis of species and size specific mortalities was carried out using a database that was compiled by Mike Kaiser and Hilmar Hinze. The results of 101 different experimental manipulations or observations of the effects of fishing disturbance on benthic fauna and communities, extracted from 55 separate publications were collected (Kaiser et al. MS), Appendix I). This database does not include comparative studies that studied areas of the seabed subjected to different levels of fishing activity, as these have an unknown level of fishing frequency and intensity. Some studies were sub-divided as they incorporated distinctly different experimental manipulations conducted under different environmental conditions, for example comparable manipulations of a fishing disturbance but in two distinctly different habitats.

Experimental studies were classified with respect to a range of variables that might affect the degree of trawling impact This included fishing gear type, disturbance regime, water depth (m), the minimum dimension of the reported scale of disturbance (e.g. the width of a trawl), habitat type (mud, muddy sand, sand, gravel and biogenic habitat), and taxonomic grouping (e.g. by phylum). For definition of the different variables we refer to (Kaiser et al. MS). The taxonomic level at which the different studies were performed varied between phylum to species.

A selection was made from the meta-database that is applicable in the approach presented here and for the North Sea situation. Therefore we limited the studies to those that have only one discrete disturbance event. Only studies that were performed in the subtidal zone, by otter trawl or beam trawl and in cold temperate latitudes were included. A further selection was made based on the reported time in days sampled after the disturbance incidence (<2 days). We did not take different levels of background disturbance into account, as this would reduce the dataset severely. Studies on both infauna and epifauna have been taken into account.

The magnitude of the response to the fishing treatment was calculated from the following equation, using the mean values for fished and unfished plots in any given study:

% difference = ((Af – Ac)/Ac) x 100

where Af = abundance in fished plots and Ac = abundance in unfished control plots. For cases in which the study involved a Before Fishing-After Fishing comparison for the same plot(s), rather than a Treatment-Control design, these data were used to calculate % difference by comparison of the prefishing treatment (Ac) with the post-fishing condition (Af).

From the selected subset of the meta-database mean mortality rates (defined as % difference in abundance) per taxonomic group were calculated.

5.1.2 Results

The selection comprised mortality estimates on 162 different taxonomic groups from 16 different studies. Values for individual orders are presented in table 5.1. In some taxonomic groups variation between studies is very large, for instance in Forcipulatida and Copepoda. In these cases this is caused by one outlying study. Generally there is a reduction in mean density after fishing, only in some studies densities were found to increase (some annelida and echinodermata). Reductions in densities are on average larger in experiments with beam trawls than with otter trawls (fig. 5.1).

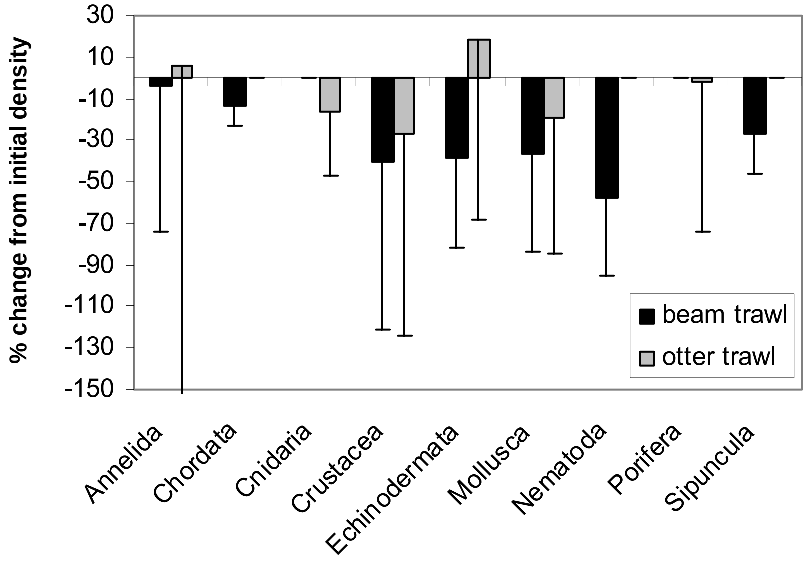


Figure 5.1. Mean mortalities (+SD) for different orders.

Table 5.1. Mortality per gear and order.

phylum	class	order	beam trawl average	SD	n	otter trawl average	SD	n
Annelida	Oligochaeta	*undefined*			0	-26.8	*28.5*	3
	Polychaeta	Capitellida	-13.5		1	-52.1	*13.3*	3
		Cossurida			0	-16.1	*13.9*	3
		Eunicida			0	-30.6	*3.2*	3
		Opheliida			0	-6.2	*36.7*	4
		Orbiniida	-58.5	*45.9*	2	-33.3	*16.0*	7
		Oweniida	70.0		1			0
		Phyllodocida	-9.7	*64.7*	11	-14.8	*37.6*	18
		Sabellida			0	-65.3		1
		Spionida	26.3	*89.4*	7	-19.7	*51.1*	24
		Terebellida	-23.0	*63.3*	5	-76.7	*2.8*	2
		undefined			0	11.0	*34.0*	4
Chordata	Ascidiacea	Pleurogona	-13.7	*9.5*	3			0
Cnidaria	Anthozoa	Gorgonacea			0	-7.5	*13.0*	3
		Valvatida			0	-14.7		1
	Ascidiacea	Pleurogona			0	40.8		1
	Hexacorallia	Actiniaria			0	-62.5		1
		Scleractinia			0	-40.0		1
		undefined			0	-30.6		1
Crustacea	Copepoda	*undefined*			0	-10.5	*151.2*	7
	Crustacea	Decapoda	-6.0		1			0
	Eumalacostraca	Amphipoda	-16.3	*125.2*	9	-46.6	*47.6*	9
		Cumacea	-31.0		1			0
		Decapoda	-62.9	*25.4*	11	72.8		1
		Mysidacea	-50.0		1			0
		Tanaidacea			0	-76.9	*25.9*	2
		Tanaidae			0	-75.5	*24.7*	2
	Ostracoda	*undefined*			0	50.8	*104.9*	2
Echinodermata	Echinoidea	Echinoida	9.1		1	-46.7	*5.0*	2
		undefined	-12.4	*33.3*	2			0
	Holothurioidea	Aspidochirotida			0	-16.8	*22.2*	2
		Dendrochirotida	-17.3	*33.3*	2			0
		undefined	-94.3		1			0
	Stelleroidea	Comatulida			0	-23.9		1
		Forcipulatida			0	158.3	*142.2*	3
		Ophiurida	-57.3	*51.0*	6	-43.3		1
		Paxillosida	-33.6	*34.5*	5			0
		Phrynophiurida			0	0.0		1
		Spinulosida			0	21.1	*5.5*	2
		Valvatida			0	22.8	*35.3*	4
		Velatida			0	-15.1	*93.9*	3
		undefined			0	-2.2		1
Mollusca	Gastropoda	Cephalaspidea			0	-19.8	*25.6*	2
		Mesogastropoda	-40.2	*36.0*	6	271.4		1
		Neogastropoda			0	108.1		1
		undefined	-81.8		1			0

Table 5.1. Continued.

phylum	class	order	beam trawl			otter trawl		
			average	SD	n	average	SD	n
Mollusca	Pelecypoda	Cephalaspidea	-20.0		1	-19.0	*25.5*	2
		Myoida	8.5	*44.1*	5	-43.1	*17.3*	4
		Nuculoida	-6.0		1	-41.8	*49.8*	6
		Veneroida	-44.5	*33.9*	51	-28.0	*37.0*	19
		undefined	66.0	*189.6*	2			0
Nematoda	Adenophorea	Chromadorida	-64.0		1			0
		Monhysterida	-100.0		1			0
	undefined	*undefined*	-33.9	*34.1*	2			0
Nemertea	*undefined*	*undefined*	-100.0		1	300.0		1
Porifera	Demospongiae	Hadromerida			0	81.8	*167.1*	2
		Haplosclerida			0	10.0	*14.1*	2
	undefined	*undefined*			0	-29.6	*25.3*	7
Sipuncula	Sipunculidea	Golfingiiformes	-27.0	*19.5*	3			0

As Kaiser et al (MS) showed, mortality rates are also highly dependent on habitat characteristics. Strictly speaking the above values should be differentiated for different habitats. However many benthos species are highly specific in their habitat preferences and generally occur in only one habitat (i.e. sand, gravel, mud). In addition, the resolution at which these data can currently be applied (ICES square) is much smaller than the resolution of habitat distribution.

5.2 Direct mortality of fish

5.2.1 Methods

For fish there are no estimates of the encounter mortality caused by the passing of the gear. In fishery science catchability is a known concept (e.g. (Dickson 1993)). It is easy to measure the amount caught but as long as there are no reliable estimates of the true abundance before the passing of the gear it is impossible to determine catchability. The main problem is that any gear that is used to determine the initial (pre-hauling) abundance will suffer from the same or similar limitations that prevented the determination of the catchability in the first place; any gear will only catch a gear-dependent subset of a population or community. To further complicate the matter, encounter mortality consists not only of mortality of animals caught in the net (i.e. catchability) but also mortality due to contact with the gear (e.g. after passage through the net).

Here we try to identify at least some of the factors involved. The interaction between fish and bottom trawls is determined by fish behaviour in relation to gear characteristics, making the catch efficiency of a gear hard to quantify (Wardle 1988) (Dickson 1993). Based on the available literature (Weinberg et al. 2002) (Engås and Godo 1989) we developed a model in which catch efficiency is determined by four factors:

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

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

5.2.2 Results

Positioning in the water column

The positioning in the water column of the fish relative to the gear determines the likelihood that fish enter the mouth of the net. As there are no quantitative data we assume that 95% of the roundfish are positioned such that they do not succeed in escaping over the headline of the otter trawl and as a beam trawl has a markedly lower vertical opening this is assumed to be only 60% for the beam trawl. Flatfish are assumed not to be able to pass over the top of both types of gear.

Herding

Fishing gear is often designed such that it guides fish into the path of the net. This effect mainly applies for otter trawls and is called herding. Herding may therefore result in an effective width of the gear larger than the actual width of the net. Not all fish species between the otter boards are herded towards the mouth of the net ((Wardle 1986); (Dickson 1993);(Engås and Godo 1989); (Ramm and Yongshun 1995). For roundfish, Engås & Godø (1989) compared the catches of cod and haddock between gears with different sweep lengths. With increasing door-spread, a significant increase was found in catches for cod and haddock, especially for larger fish lengths (Engås and Godo 1989). Herding is assumed to be related to the door-spread of the gear. From Engås & Godø (1989) we used an average herding effect per meter door-spread of 0.067. We assumed a standard otter trawl has a sweep-length of 40m, a door-spread of 58m, a net opening of 19m, and hence a difference of 39m in width between door-spread and net opening (Engås and Godo 1989). As we assumed a effective width of the gear equal to the doorspread (i.e. 58 m) the correction factor for the proportion of fish that do not reach the mouth of the net is calculated as $(39+19*0.067)/58=0.69$. No quantitative data on herding were found for flatfish. Although it is generally accepted that the herding of fish by trawl sweeps is size-and temperature dependent (Wardle 1983{Winger, 1999 #556}}, there are no direct empirical studies that show this. Because of this lack of quantitative estimates we assume no herding effects for flatfish in the otter trawl. The beam trawl is not designed to herd the fish and the effective width is assumed equal to the width of the beam.

Escape below footrope

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

Retention in the net

Most fish are considered to escape from the cod-end of the gear (Millar and Fryer 1999) and therefore most studies on gear selectivity have been carried out on cod-end selection (Wileman

et al. 1996). Gear characteristics such as mesh size, cod-end extension length, cod-end diameter or mesh-shape have a significant influence on the selection of fishing gears (Beek et al. 1981, 1983),(Reeves et al. 1992); (Robertson et al. 1988); (Zuur et al. 2001). The proportion of fish that is retained in a net is calculated as a function of mesh size using cod-end selectivity data. (Wileman 1991) summarized several gear selectivity studies carried out over a period of more than 30 years. Several species in two types of gear were distinguished: seven species in the otter trawl (OT) and two in the beam trawl (BT) (table 5.2).
A logistic curve is used to describe the relationship between the length of a fish and the proportion of a population that is retained in a net (Casey 1996).

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

Where:
S_L = The proportion of the population of length L and class width Δ L that is retained.
L50 = The length of which 50 percent of the population entering the net is retained (cm)
L25=The length of which 25 percent of the population entering the net is retained (cm)

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

L50 = SF * M
L25 = L50-(SR/2)

Where:
SF= Selection Factor
M= Mesh size (cm)
SR=Selection range (cm)As sufficient quantitative information to determine cod-end selectivity is only available for some commercial species (Maclennan et al. 1992) we determined selectivity parameters for roundfish and flatfish and applied those to the non-target species.

The values for the positioning, herding and footrope (small/large fish) factor are assumed constant (table 5.3). These factors are multiplied to result in a final efficiency factor. Thus a beam trawl is more selective than an otter trawl for flatfish (1 versus 0.12 for small and 0.21 for large flatfish) and less selective for roundfish (0.6 versus 0.77 for roundfish) (table 5.3).

Table 5.2. Gear selectivity parameters: selection factor and selection range for different species and species groups. Two types of gear have been used. OT=Otter trawl, BT=Beam trawl. Mean values for roundfish and flatfish species have been calculated. Note that the mean value for flatfish does not include sole.

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

Table 5.3. Factors used in the direct effect model for calculation of catch efficiency for beam trawl (BT) and otter trawl (OT) and different fish types, demersal roundfish (DR), demersal flatfish (DF) and pelagics (P). The factor is dependent both on fish-size and mesh-size. The footrope factor is divided in a factor for smaller (S, < 25 cm) and a factor for larger (L, ≥ 25 cm) fish. The overall factor (S,L) is calculated by multiplying the positioning, herding and footrope factor.

	Fish	Factor					
Gear	type	Positioning	Herding	Footrope (S)	Footrope (L)	Overall (S)	Overall(L)
BT	DR	0.60	1.00	1.00	1.00	0.60	0.60
BT	DF	1.00	1.00	1.00	1.00	1.00	1.00
BT	P	0.05	1.00	1.00	1.00	0.05	0.05
OT	DR	0.95	0.85	0.95	0.95	0.77	0.77
OT	DF	1.00	0.30	0.40	0.70	0.12	0.21
OT	P	0.25	0.85	0.95	0.95	0.20	0.20

5.3 Discussion

5.3.1 Direct mortality

Limitations of the available data forced us to make a large number of assumptions to be able to calculate the mortality estimates of fish. We only distinguished two types of gear, beam trawl and otter trawl neglecting the large variation in rigging within these types. The catch efficiencies of these two gears that were applied to all species were based on experiments conducted on only a few commercial species. For their interaction with the gear non-commercial species were characterized by their size and shape (i.e. flatfish or roundfish). Also it was assumed that fish do not redistribute after an area is trawled. This will underestimate mortality. Effort and therefore mortality is also probably slightly underestimated because not all countries that have a trawl fishery in the North Sea are represented in the data (Jennings 2000).

The meta-database of mortalities of benthos in response to fishing contains a wide variety of studies. Due to the strict requirements of our approach on the input data only a small subset of the database could be used. As explored in (Kaiser et al. MS) the response of benthos to fishing disturbance is highly dependent on habitat. Therefore habitat-specific mortalities should be used in further calculations. A problem with most studies in this database is that the research areas have already been disturbed by fishing for a long time. Therefore it is only possible to study the effects of fishing against a certain background disturbance (that may greatly differ in intensity between studies).

5.3.2 Mortality of fish that have escaped from bottom gear

In the approach described we do not yet take into account the consequences of the injuries of fish that have been in contact with the gear. Studies that have investigated this are scarce, not so recent, and only deal with the short term effects. Longer term effects such as predation on injured fish and the ability to fully recover from injuries or stress are limited (Chopin and Arimoto 1995, Mellegaard and Bagge 1998). Chopin and Arimoto (1995) present an overview of available studies up to 1995. Mortalities of fish escaping from bottom trawlers measured in various experiments are highly variable (table 5.4). Also mortality seems to be related to size (Sangster and Lehmann 1993, Ingolfsson et al. 2002). Experiments to measure mortality of fish that escaped through the codend differ greatly in their approach and therefore the resulting mortalities are not directly comparable. The reliability of estimating mortality using a codend cover (as most studies do) has also been questioned (Breen et al. 2002), because the reduction of water flow induced by the cover around the codend is likely to affect escape behaviour and reduce the likelihood of injury during escape. Besides the flow of water inside the codend is still so high, that the majority of smaller fish escaping from the codend are unable to sustain swimming speeds demanded by this water flow.

In effect the mortality estimates derived according to the method described here are likely to be higher if the reduced mortality of escaped fish are taken into account as well.

Table 5.4. Mortalities of fish escaping from bottom trawls (after (Chopin and Arimoto 1995) with added sources).

gear	species	mortality (%) min	max	reference
otter trawl	striped bass	1	16	(Hislop and Hemmings 1971)
otter trawl	haddock	9	27	(Sangster and Lehmann 1993)
otter trawl	whiting	10	35	(Sangster and Lehmann 1993)
otter trawl	gadoiids	14	100	(Main and Sangster 1990)
otter trawl	cod	0	0	(Soldal et al. 1991)
otter trawl	haddock	1	32	(Soldal et al. 1991)
otter trawl	atlantic halibut	65	65	(Nelson et al. 1989)
otter trawl	scup	0	50	(DeAlteris and Reifsteck 1993)
otter trawl	flounder	0	15	(DeAlteris and Reifsteck 1993)
otter trawl	cod	0	0	(DeAlteris and Reifsteck 1993)
otter trawl	red mullet	17.1		(Lok et al. 2002)
otter trawl	sea bream	0	0	(Lok et al. 2002)
otter trawl	cod	3		(Suuronen et al. 2002)
otter trawl	haddock	26	50	(Ingolfsson et al. 2002)
shrimp trawl	young gadoids	0	0	(Soldal and Engas 1997)

6. Case study for impact of fishing on non-target fish

To illustrate the approach presented in chapter 2 we summarize the results of a study in which the direct effects model was developed for impact of fishing on non-target fish species (Piet et al. MS-b).

6.1.1 Absolute abundance

The method to estimate the abundance of non-target species is described in chapter 3.

6.1.2 Trawling frequency

The frequency with which an area is trawled is considered to be a better measure of fishing impact than conventional effort measures such as days-at-sea or hours fished. Piet et al. (submitted) provides quantitative data on relevant fishing parameters (e.g. the proportion of the day actually spent fishing, fishing speed) and gear characteristics (e.g. width of the gear) that allow the transformation of these conventional measures into trawling frequencies. For fishing effort we used the data of Jennings et al. (1999a and 2000), on the international otter- and beam trawl effort for the period 1990-1995. Trawling frequency (F_t) is calculated as:

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

Where:

F_t = Frequency trawled
Eff_w =Effective width (m)
T_F =Time Fished (s)
S=Speed (m/s)
S_{ICES}=Surface of ICES rectangle (m^2)

6.1.3 Impact of the gear: mortality

The method to arrive at size and species-specific mortalities is described in chapter 5. The DEM was used to calculate the fishing mortality caused by beam trawl and otter trawl for all fish species. The data presented here are still preliminary and will be adjusted as the method is developed further.

Although the mean species-specific mortality of non-target species (23%) is well below that of commercial species (46%), the range of mortalities is wider varying between 0-96% as opposed to 19-67%.

Size-specific mortalities of four groups of demersal non-target species are shown in figure 6.1. According to this model the fishing mortality increases with increasing size of the fish, first for the flatfish and rays followed by the roundfish and sharks. Mortality of the larger size-groups is highest for the fish at about 69% for flatfish to 75% for roundfish and lower for the elasmobranchs at about 64% for the sharks and 43% for the rays. Pelagics still suffer about 10% mortality.

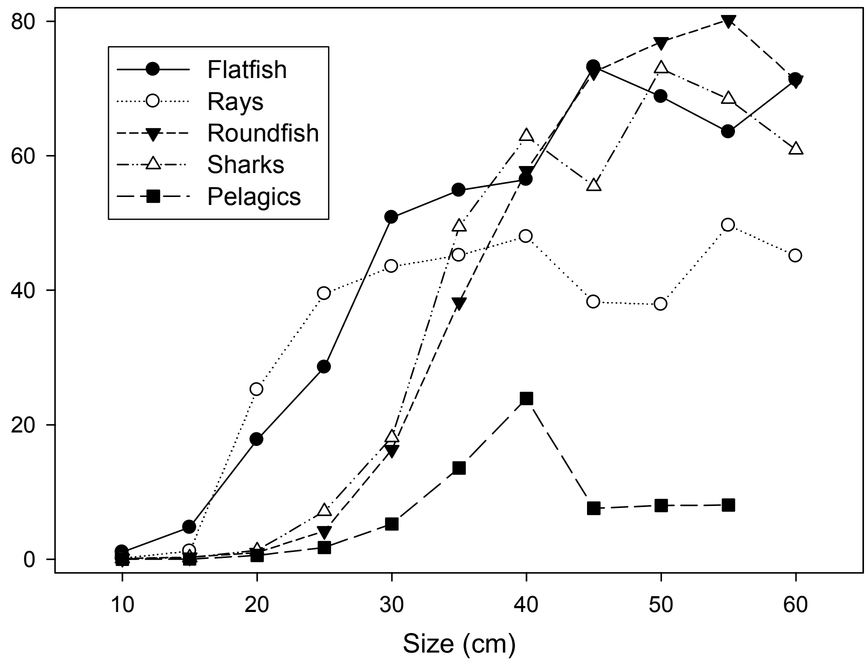


Figure 6.1. Estimates of mortality caused by trawling of different components of the demersal fish community.

6.1.4 Conclusion

The effects of bottom trawling on the demersal fish community are considerable with large differences in mortality between species due to differences in size and how their distribution overlaps with that of the fishery. For species such as elasmobranchs which, due to their life-history characteristics (slow-growing and late maturing) are known to be extremely vulnerable to fishing mortality, alarmingly high mortality rates were found that are not sustainable.

Although knowledge on notably the impact of the gear and availability of data may affect the accuracy of the outcomes of this model this approach appears useful to assess the effects of bottom trawling specifically on the non-target species.

7. Potential indicators of fishing disturbance

Indicators of fishing pressure are necessary to support an ecosystem approach in fisheries management. Depending on the information available it is possible to develop pressure indicators that are more suited to describe the actual pressure on the ecosystem and its components. We present a framework that distinguishes six levels of increasingly better pressure indicators that also require increasingly more information to be quantified (fig. 7.1). We use the example of the Dutch beam trawl fleet in the North Sea to evaluate the performance of these pressure indicators at different levels of information content.

The first level is that of fleet capacity (i.e. number of vessels). The second level is effort often expressed in days-at-sea. The third level incorporates fishing parameters like the proportion of time actually spent fishing, fishing speed or gear characteristics such as the size of the gear in order to determine the frequency with which an area is fished. The fourth level also takes the spatial distribution of effort into account. Ultimately, fishing pressure is best described by the fishing-induced mortality. This is calculated at the 5th level through the introduction of the impact of the passing of the gear on an ecosystem component. Finally at the highest level the spatial distribution of this ecosystem component relative to that of the fishery is used to include the probability of encounter into the mortality. For details on these indicators we refer to appendix 2.

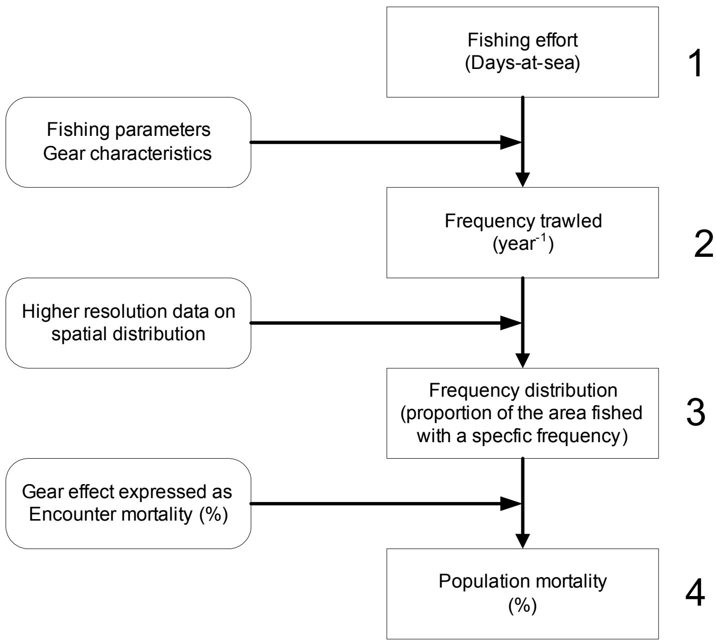


Figure 7.1. The approach for developing indicators of fishing disturbance at different levels of information content. The boxes on the left describe the type of information required, the level is indicated to the right. Encounter mortality is the % mortality caused by the singular passing of a specific type of gear.

8. Acknowledgements

We would like to thank Mike Kaiser and Hilmar Hinze for being able to use the meta-database and Leonie Robinson for help in making the proper extractions.

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Appendix 2. Potential indicators of fishing disturbance

Manuscript by
G.J. Piet, F. Quirijns, L. Robinson and S.P.R. Greenstreet

Introduction

When implementing an ecosystem approach in fisheries management, indicators are required (1) to describe the pressures affecting the ecosystem, the state of the ecosystem and the response of managers (2) to support management decision making, (3) to track progress towards meeting management objectives and (4) to communicate the effects of complex impacts and management processes to a non specialist audience.

Many indicators are proposed to support an Ecosystem Approach to Fisheries Management (EAFM) but few (if any) indicators that track changes in the marine environment can support management (Rice, 2000) as causes of changes may not be known or manageable and because indicators do not provide guidance for managers if limits, targets, reference trajectories or directions cannot be set. Indicators required to support an EAFM not only need to provide information on the state of the various ecosystem components, but also on the manageable activities that affect these ecosystem components and that the link between these two needs to be thoroughly understood (Daan, 2005). Several methods have been proposed for classifying environmental management indicators on this basis, and a widely used framework is the pressure state response (PSR) system (Garcia & Staples, 2000). This framework uses pressure indicators (P) to measure the pressure impacting an ecosystem component, state indicators (S) to measure the state of the ecosystem component and response indicators (R) to measure the response of managers to the change in state. Since policy commitments and associated objectives relate to state, the reference points, trajectories or directions needed to measure progress towards meeting objectives are initially set for state indicators. Achievement of these reference points, trajectories or directions will, by definition, mean that the operational objectives are being met. Once reference points, trajectories or directions have been set for state indicators, and the links between pressure, state and response are known, then corresponding reference points, trajectories or directions can be set for pressure and response indicators. Pressure and response indicators are essential in this process as they often have the desirable properties of ease of measurement and rapid response times. As a result, guidance for year on year management decision making is often better based on pressure and response indicators, with changes in state assessed less frequently to confirm that pressure and response are affecting state as predicted (Nicholson & Jennings, 2004). Pressure indicators therefore will need to be able to show how the following management measures affect fishing impact:

- Effort control through decommissioning or days-at-sea
- Technical measures aimed at increasing the selectivity of gears using separator devices, escape hatches, increased mesh-size or pingers to reduce by-catch
- Protected areas which can be created by permanent or temporary closures of areas to certain types of fishing

The ecosystem components that are most directly affected by fishing are fish and benthic invertebrates (For reviews see (Collie et al., 2000; Dayton et al., 1995; Hall, 1999; Jennings &

Kaiser, 1998; Kaiser & de Groot, 2000) and many indicators have been proposed that describe the state of these components at different hierarchical levels (e.g. population and community). However, the link between the state of these ecosystem components and a manageable activity (i.e. fisheries) is often unclear as most studies are correlative and descriptive, examining the relationship between a difference in the 'level' of fishing effort and a particular response, such as a change in species diversity, size spectra or species composition of the community (Piet & Jennings, 2005). 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 ecosystem components are structured by a combination of biotic (e.g. competition, predation and larval dispersal) and abiotic factors (e.g. climatically driven changes in temperature and productivity) (Murawski, 1993; Clark & Frid, 2001; Kröncke & Bergfeld, 2001) which have in common that they affect these components through induced mortality, either directly or indirectly. In theoretical ecology terms, disturbance is the mortality caused by perturbations to the ecosystem. Thus, ultimately fisheries disturbance should also be considered as an anthropogenic source of mortality.

The development of Pressure indicators for fisheries has been hampered by confusion over 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, i.e. direct and indirect effects. However, when considering disturbance within theoretical modeling 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 response of ecosystem components to fisheries disturbance one must first establish the level of mortality experienced by these components before inputting this to an overall model of the factors that structure them.

The main objective of this paper is, therefore, to explore how Pressure indicators can describe the actual disturbance caused by fishing (i.e. destruction of habitat, mortality of ecosystem components), what type of information is needed to quantify these indicators and how they are relate to different types of management measures. Here, we will present pressure indicators that perform better at describing fishing disturbance as the information content of them is increased and ultimately result in a measure of direct mortality. This implies that at this level pressure indicators can be directly linked to several of the existing state indicators and more realistic reference levels can be set. We will use the North Sea as a case study to exemplify the differences between the potential pressure indicators at different levels of information content.

In the North Sea, Dutch vessels are responsible for more than 70% of the beam trawling effort (Jennings et al., 2000). Because of its mode of operation (dragged over the seabed) beam trawling probably causes the largest unwanted ecosystem effects of fishing. Yet, in the North Sea beam trawling effort is still reported using some of the least informative measures of effort (i.e. days-at-sea) at a spatial scale of ICES rectangles (approximately 30x30 Nm) with no accounting for how factors such as the encounter mortality of the gear on the species or the spatial (micro-) distribution of effort will cause variation in the mortality induced. (Piet et al., 2000) showed for a number of benthic species how these factors affect population mortality estimates.

In this study we introduce a framework that identifies potential pressure indicators at different levels of information content. We will use the Dutch beam trawl fishery in the North Sea and a virtual population to illustrate the performance of these pressure indicators as measures of disturbance that are representative of the actual impact this type of fishing has on the ecosystem and its components. The consequences of these findings for the collection of data needed to support an EAFM will be discussed.

Methods

The ecological disturbance of fishing is described at levels of increasing information content (Figure 1). The bottom level is that of fleet capacity, i.e. the number of vessels in a fishery where different métiers may be distinguished. The next lowest level is that of fishing effort, calculated as fleet capacity (usually in numbers but this may also be in tonnage or engine-power) multiplied by their activity, here expressed in days-at-sea. Information on fishing practices and gear characteristics allows the calculation of the third level indices: the frequency at which the seabed was swept or a volume of water trawled. The impact of fishing on a habitat, fish- or benthic ecosystem components is not only determined by the measure of effort (e.g. days-at-sea or frequency) but also how this effort is distributed within that area (both currently and in relation to the historic distribution of effort); an even distribution of effort will have a bigger impact than that same amount of effort concentrated on a relatively small area, leaving the remainder unaffected. Depending on the data available the spatial distribution can be ignored (Level 3) or incorporated at different spatial resolutions (e.g. Level 4 Low or High). Finally, by combining the 4^{th} level indices with information on the effects of the gear and the distribution of the ecosystem components we reach the highest level indicators (respectively level 5 and 6) which actually give the fishing mortality of a (group of) ecosystem component(s). As this mortality can be determined from any of the level 3, Level 4 Low and Level 4 High configurations we distinguish similar configurations for Level 5 and 6 (i.e. None, Low, High). Each of the types of information required (see Figure 1) will be exemplified using data from the Dutch beam trawl fleet in the North Sea.

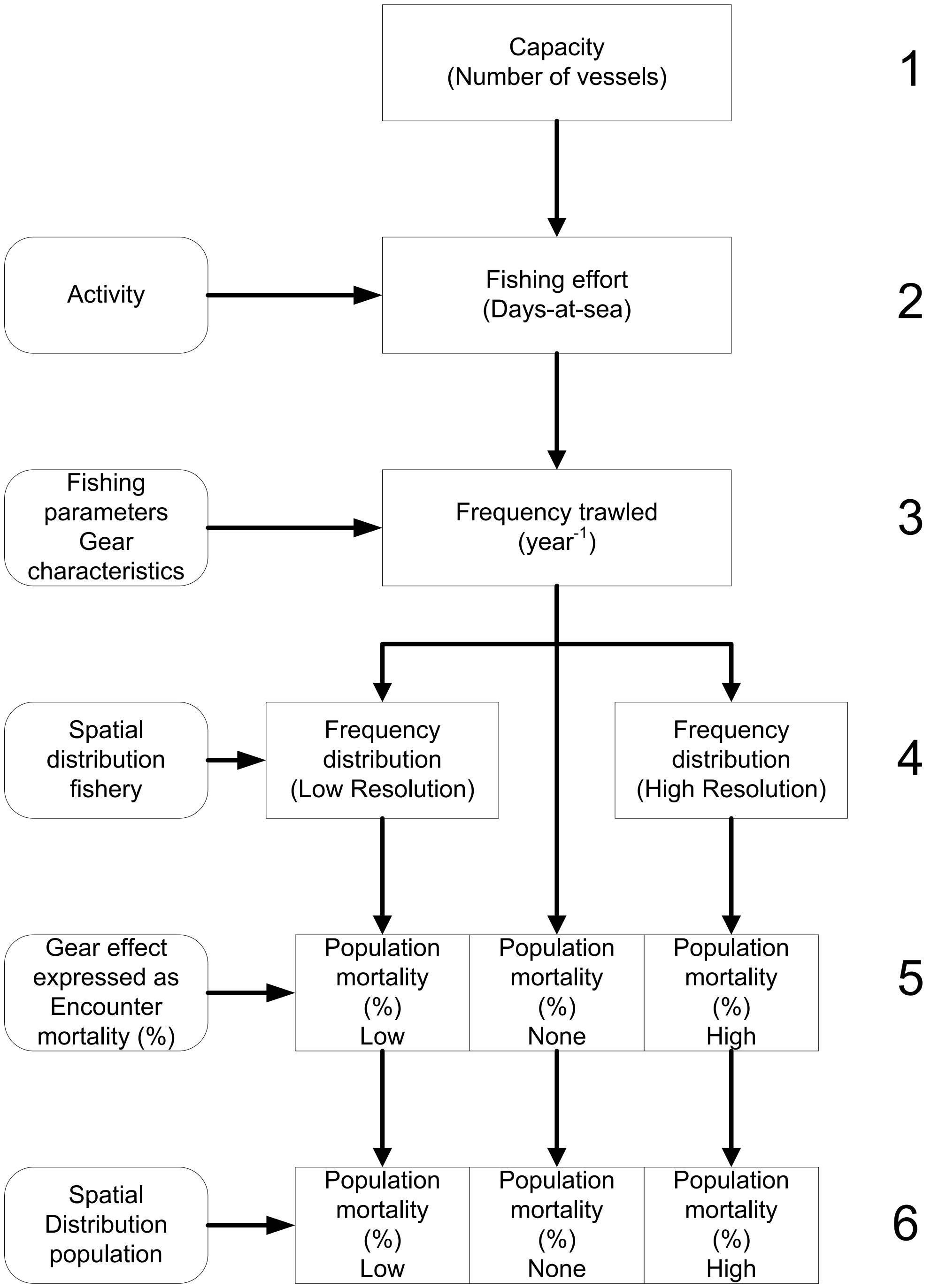


Figure 1. Schematic representation of Pressure indicators at different levels of information content.

Levels 1 and 2: Fleet capacity and Fishing effort

Data on the number of vessels, their activity and how this is distributed spatially at a resolution of ICES rectangles (30x30 Nm) is available from the EC-logbooks of the total Dutch fleet (VIRIS data base). The database distinguishes different segments of the fleet based on their engine-power, contains information on the time of the start and end of the fishing trip, the gear used, the ICES rectangle fished and the landings by fish species. The database is designed for quota management purposes but available for research purposes and similar databases are available for other EU countries.

Fleet capacity is expressed as the number of Dutch beam trawl vessels per segment, fishing effort as the number of days-at-sea per segment.

Level 3: Frequency trawled

Fishing activities in a specific area cause a proportion of that area being fished with a mean frequency. From the VIRIS database it can be determined which ICES rectangles are fished and from that the proportion of the North Sea area covered by the fishing fleet can be calculated as the sum total of the surface area of the ICES rectangles divided by the total North Sea area available for trawling where the latter was defined as ICES area IV minus the part deeper than 200m. All areas were calculated using GIS (projection UTM-1983, zone 31). The mean frequency per year was calculated as the total area trawled by the Dutch fleet divided by the total area of the ICES rectangles fished.

To determine the total area trawled fishing effort in days-at-sea needs to be multiplied by the area trawled per day at sea. This is determined by a number of fishing parameters together with gear characteristics. For the Dutch beam trawl fleet these are largely determined by engine power. Therefore two segments are distinguished in the Dutch beam trawl fleet based on engine power: (1) eurocutters with engine-power below 300 Hp and (2) large vessels with higher engine-power. Three datasets were used to determine the fishing practices of each of these segments; One based on logbooks of the fishermen where they register for each tow the time set and hauled together with the trawling speed, the others based on micro-scale distribution data collected using APR (Automated Position Registration) or VMS (Vessel Monitoring through Satellite). A sample of about 10% of the Dutch beam trawl fleet was equipped with APR for the period 1993-2000 (see (Rijnsdorp et al., 1998). VMS data became available from 2000 onwards when positions of all EU vessels >24 m were recorded for enforcement purposes. For a proper assessment of the disturbance caused by fishing it is essential to be able to distinguish fishing registrations from other activities (e.g. steaming, laying still). This distinction can be made using the speed of the vessel. As the logbook data provide the speed at which fishing took place, the fishing speed-range can be determined from those. The speed in knots (=$Nm.hr^{-1}$) was determined from the APR data by calculating the distance between subsequent positions and dividing by 0.1 hour (= time interval of 6 minutes). The VMS data usually provide the measured speed at each position. The proportion of time spent fishing can be calculated as the ratio of the number of registrations within the fishing speed range and the total number of registrations.

The frequency with which an area is trawled was calculated as:

$$F = E * HF * S * 1852 * 2 * W / A$$

Where *E*=Effort (days-at-sea), *HF*=hours fished, *S*=speed in knots, *W*=width of the beam (m) with two beam-trawls per vessel and *A* = surface area (m^2)

Level 4: Frequency distribution

The spatial distribution at low resolution (i.e. Level 4 Low) can be determined using the VIRIS database where the spatial resolution is at the level of ICES rectangles (approx. 30x30 Nm). To describe the distribution of trawling frequencies at a high resolution (i.e. Level 4 High), we used the APR/VMS data for the period 1994-2004. For the micro-scale distribution we used a spatial scale of 1x2 minute squares (approximately 1x1 Nm) as Rijnsdorp *et al.* (1998) found that within these squares the distribution was random. One ICES rectangle consists of 900 (30x30) of these squares. In order to combine APR and VMS data with different time intervals between registrations, the unit of effort of the APR and VMS data is registrations standardized to a 1-hour interval. The speed of the vessel was used to select fishing registrations. Similar to what was described at a resolution of ICES rectangles we can determine the proportion of the area that was fished. The APR/VMS data, however, allow a distinction between:

- the proportion that is not fished because it is outside the range of this fleet or because of obstacles such as oil rigs or wrecks. This is comparable to the un-fished proportion at level 3 (i.e. ICES rectangles with no effort attributed to them) and calculated similarly using all squares except those with at least one registration over the entire sampling period;
- the proportion that is not fished for other reasons (e.g. low yield or unsuitable ground). This was calculated by considering every square with at least one registration over the entire sampling period a “fishable” square and each year the proportion of un-fished squares was based on the “fishable” squares only.

Next, the frequency with which a square was trawled was calculated similar to the low-resolution frequency based on logbooks (VIRIS) but instead of $E*HF$ we used FR which is the number of standardized fishing registrations.

If the APR/VMS data are only available for a subset of the fleet a raising factor needs to be applied equal to the ratio between total effort of the fleet and the effort of the subset for which APR/VMS data are available (see Rijnsdorp et al. 1996). Likewise, if the total number of registrations is used instead of fishing registrations only, a correction factor for the proportion of fishing registrations in the total number of registrations needs to be applied. This could be a correction factor per segment of the fleet but may also be determined for each spatial unit.

Level 5 and 6: Population mortality

Ultimately the effect of a fishery on an ecosystem component needs to be expressed as population mortality and this depends on (1) the chance of encounter in a spatial unit between a unit of fishery and a unit of the ecosystem component which is determined by the overlap in distribution and (2) the impact this encounter has on the component, here expressed as encounter mortality (defined as proportion mortality caused by the singular passing of the gear). The encounter mortality depends on characteristics of the population (e.g. size, position in the water-column or sediment, fragility or swimming speed) in relation to gear characteristics (e.g. type of gear, mesh-size) (Casey, 1996; Collie et al., 2000; Wardle, 1988). The transition of the lower level pressure indices (Level 3, Level 4 Low, Level 4 High) to population mortality at level 5 and 6 was based on each of these configurations and virtual populations that differed in encounter mortality and spatial distribution.

The level 5 population mortalities were calculated according to (Piet et al., 2000) assuming a variable gear-effect, expressed as encounter mortality (e.g. 20% and 80%) caused by the singular passing of the gear. For level 5 the population is assumed to be evenly distributed over the entire area.

The level 6 population mortalities are calculated in the same manner as level 5 but also take into account the spatial distribution of the population. For this we distinguished between populations that mainly occurred in the heavily fished ICES rectangles (frequency > mean) or in the least fished rectangles (frequency < mean). Within rectangles the population is evenly distributed.

Finally the pressure indicators at different levels of information content were compared in terms of their absolute value and the change over time. For the absolute value the value in the year 2000 was chosen which was calculated as the intercept of the regression, the relative change (%) is the change per year relative to this value.

Results

Level 1: Fleet capacity

According to the VIRIS database the registered number of beam-trawl vessels declined in the last decade from 374 in 1995 to 210 in 2004 (Figure 2). The proportion of vessels with engine-power > 300 increased from 56% in 1995 to 64% in 2004.

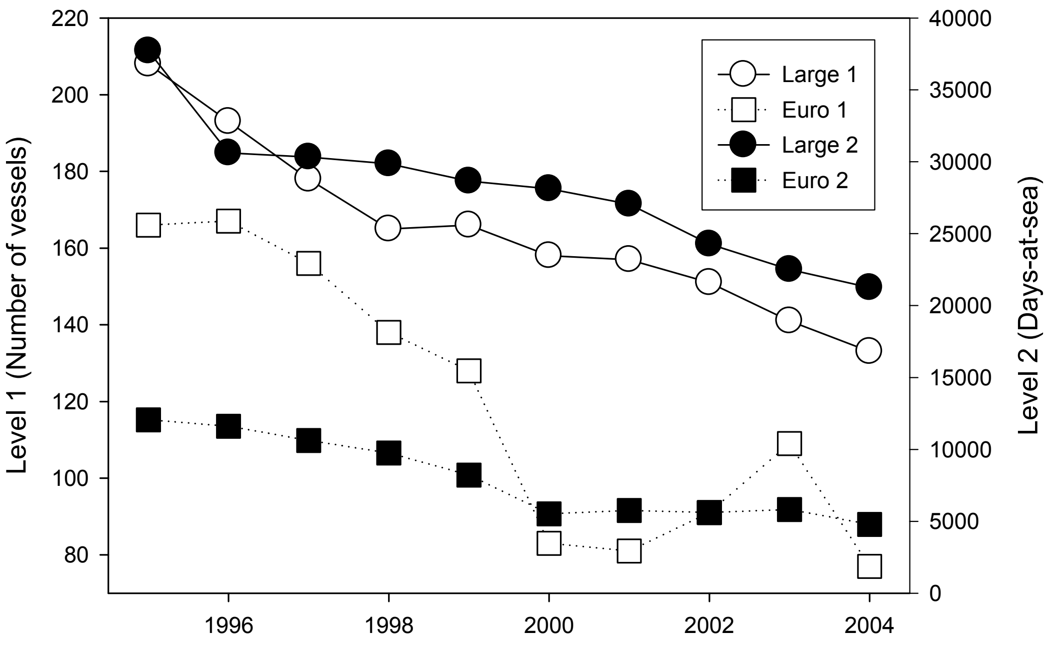


Figure 2. Pressure indicator values at level 1 and 2 (see figure 1) for two segments of the Dutch beam trawl fleet: Large vessels (engine power > 300 Hp) and eurocutters (engine power ≤ 300 Hp).

Level 2: Fishing effort

Based on the VIRIS database the activity per vessel varies considerably (Figure 3). 87% of the large vessels (≥300 Hp) spend 150-250 days-at-sea per year, with an average of 170 days-at-sea. For the eurocutters the mean activity is much less with an average of only 67 days-at-sea per year. But this is because 25% of the vessels registered less than 10 days-at-sea per year, of which many only 1 day-at-sea. This is caused by the fact that this category comprises many non-commercial vessels as well. For both segments of the fleet, mean activity per vessel per year decreases by about 1.5 days per year. Dutch beam-trawl effort decreased from 49765 days-at-sea in 1995 to 26034 days-at-sea in 2004. In the same period the proportion of fishing effort by large vessels increased from 76% to 82% (Figure 2). The spatial distribution of fishing effort at a low spatial resolution (i.e. based on VIRIS) is shown in figure 4.

Level 3: Frequency trawled

The two segments of the Dutch beam trawl fleet cover different areas in the North Sea and differ markedly both in fishing practice and gear characteristics. Usually the eurocutters deploy two beam trawls of 4 m width while the larger vessels deploy two beam trawls of 12 m width. As effort is expressed in days-at-sea we need estimates of the hours per day these vessels are fishing and the speed at which they fish in order to estimate the area swept. From the two high resolution data sources (i.e. based on APR and VMS) we determined the proportion of registrations with a certain speed, from the logbook data the proportion of hauls with a certain speed (Figure 5). From these figures we established a fishing speed range of 3-6 knots for the eurocutters and 5-8 knots for the larger vessels. Using these ranges the calculated fishing speed and proportion of the time spent fishing did not differ much between the datasets. We assumed the VMS-based estimates were the most reliable and therefore a unit of 1 day-at-sea of a eurocutter disturbs an area of 1.2 km^2 while a larger vessel disturbs 5.3 km^2. As the surface of an ICES rectangle is approximately 30x30 Nm 1000 days-at-sea of a eurocutter is equivalent to a frequency of 0.4 yr^{-1} while for a larger vessel this is 1.7 yr^{-1}. The composition of the Dutch beam trawl fleet in terms of the relative importance of the two segments varies considerably both between years and ICES rectangles which may affect the time-series or spatial distribution at this pressure indicator level compared to that at the level below. On average the eurocutters make up about 22% (period 1995-2004) of the fleet in terms of fishing effort but in the inshore rectangles and Plaice box area the fleet consists entirely of eurocutters while the large vessels dominate the offshore rectangles.

According to the VIRIS database for the period 1993-2004, about 56% of the North Sea was fished by the Dutch beam trawl fleet prior to 2001 and about 52% afterwards (figure 5). In the same period the mean frequency per year with which this area was fished decreased more gradually from about 0.68 to 0.44 $year^{-1}$ (Figure 6).

According to the APR/VMS database, 50% of the North Sea was covered by the subset of the Dutch beam trawl fleet in the database over the entire period 1993-2004. However, on an average year during this period only 26% of the North Sea surface was covered by some activity of the subset of the Dutch fleet and only 20% of the North Sea surface was actually fished (Figure 6). The time-series of the frequency shows a similar pattern to that based on VIRIS with a sudden increase in 1990 and a decrease in 2001 but the mean frequency is markedly higher varying between approximately 2 and 1.6 $year^{-1}$.

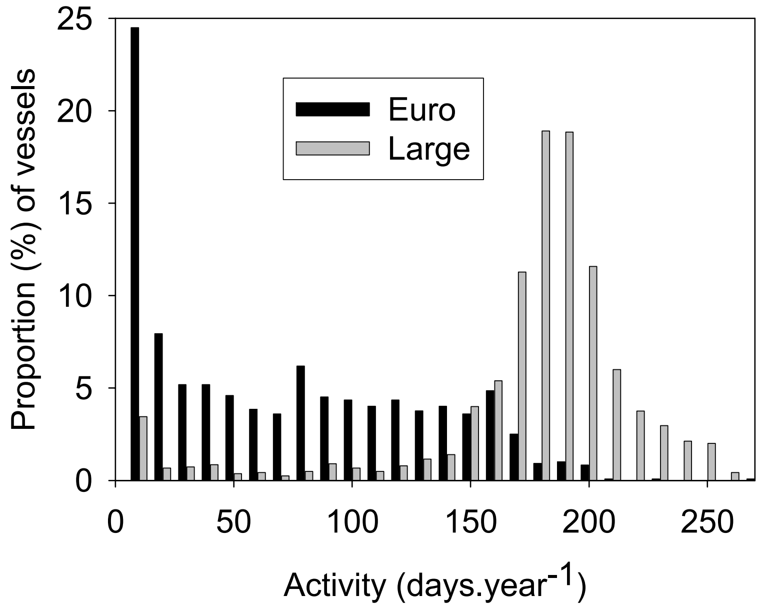


Figure 3. Activity of vessels per segment of the fleet

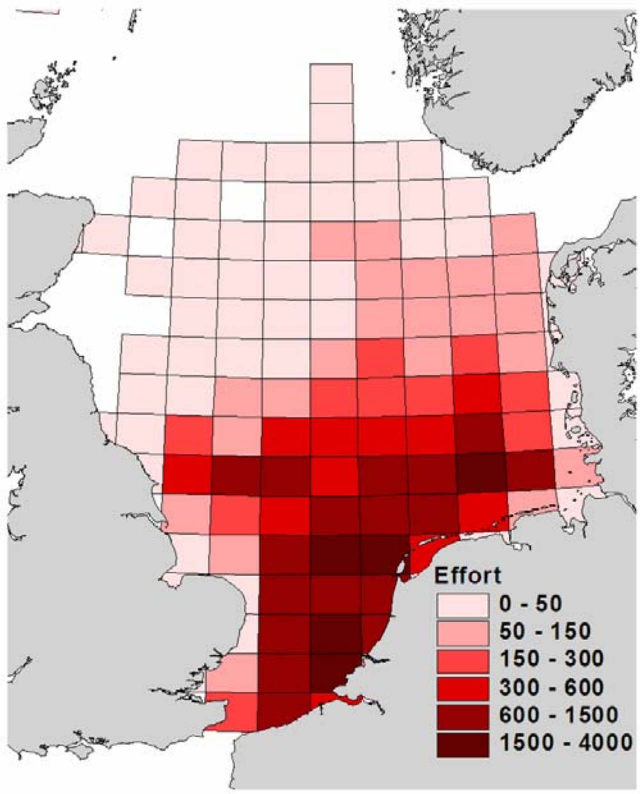
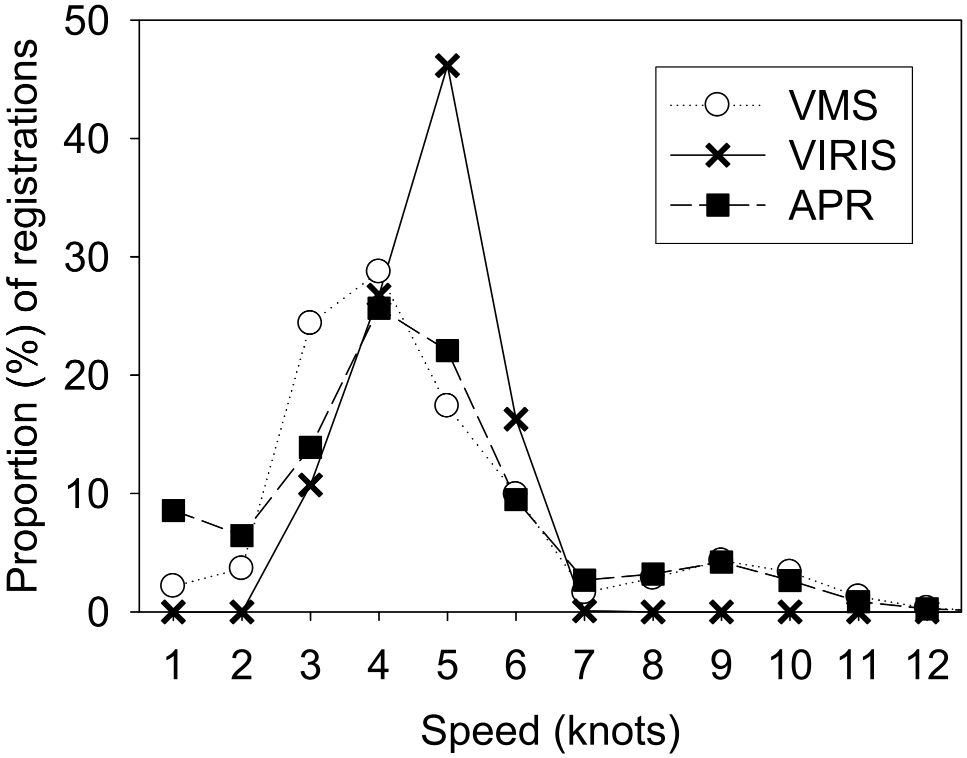


Figure 4. Spatial distribution of effort in days-at-sea according to VIRIS.



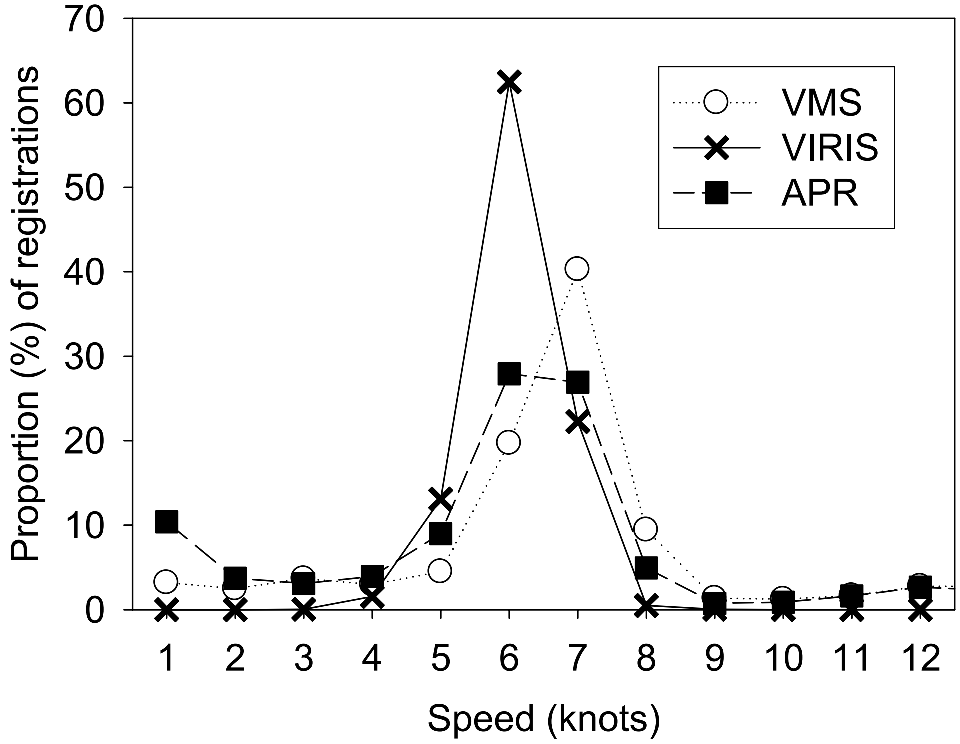


Figure 5. Trawling speed per segment based on different sources of data

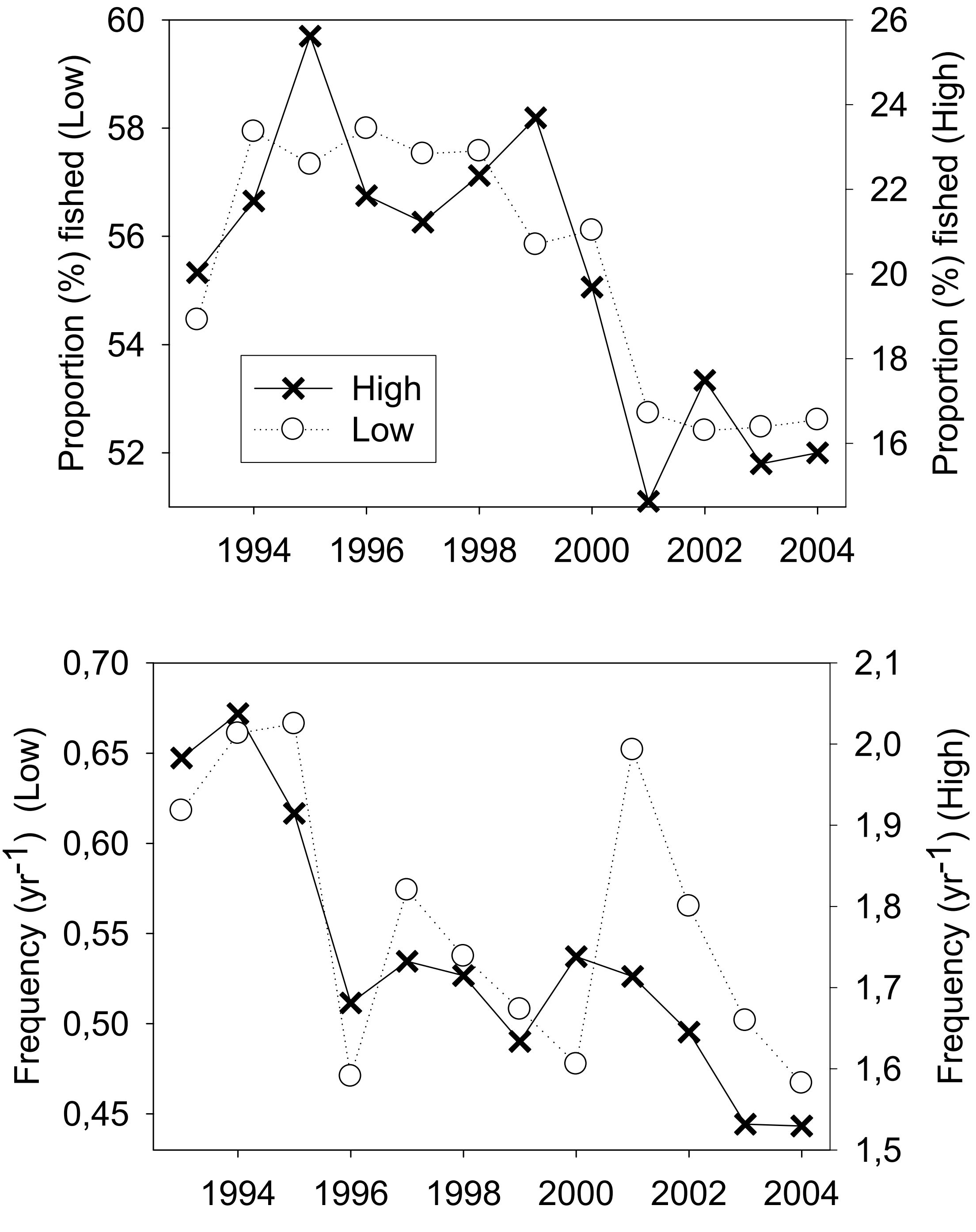


Figure 6. Indicator values at level 3, proportion of the area fished and mean frequency per year, based on low en high resolution datasets.

Level 4 Frequency distribution
The difference in frequency distribution between the VIRIS and APR/VMS datasets is reflected in figure 7 showing the occurrence of spatial units with specific trawling frequencies. Frequencies above 20 $year^{-1}$ were only observed for the micro-scale data. The frequency distribution based on VIRIS is based on just over one hundred ICES rectangles per year, for that based on APR/VMS this is based on approximately 26000 squares per year. The actual spatial distribution of fishing is shown in figure 8. The activities other than fishing (e.g. steaming, figure 9) that can be distinguished based on speed, show markedly different spatial patterns.
The degree to which the subset of the Dutch bottom trawling fleet for which APR/VMS data are available is representative of the entire fleet differs considerably between the period when APR data were used and that when VMS data became available (Figure 10). In the first period (≤ 2000) mainly large beam trawlers (15-24) and a few eurocutters (1-6) were included in the sample. From 2000 onwards this was increased to 17-37 eurocutters and 66-143 large vessels.

Level 5 and 6: Population mortality
Time-series of population mortality were created for different virtual populations and depending on the trawling frequency and its spatial distribution (Figure 11). If there is no information on the spatial distribution of effort the population mortality is only determined by the encounter mortality and the time-series are identical for populations at Level 5 or Level 6 for “low” and “high” spatial distributions. At level 6 the estimated population mortality at a “low” spatial resolution (i.e. ICES rectangles) increases considerably depending on the distribution of the population. At an encounter mortality of 20% it increases from an average of about 2% at a “low” distribution to about 25% at a “high” distribution. For an encounter mortality of 80% this is respectively approximately 5% and 80%. The population mortality at a “high” resolution (i.e. 1x1 Nm squares) is markedly less affected by either the encounter mortality or the distribution of the population. This would probably be different if the spatial distribution of the population was also determined at a high spatial distribution.

Comparison of the indicators shows that they differ considerably in their representation of the pressure on the ecosystem and its components. The absolute values of the indicators at each levels differed at 276 vessels (level 1), 34829 days-at-sea (level 2) or 0.51 $year^{-1}$ (table 2). At level 5 the population mortality varied between 6 and 30% while at level 6 this was between 2 and 73% for encounter mortalities of respectively 20% and 80%. Practically all indicators showed significant trends over time. The lower level indicators show a decrease over time (1995-2004) of almost 7%. At level 3 this is only 2.6% while at level 5 and 6 this is 7.6% when the spatial distribution of the fleet is not taken into account. When the spatial distribution of the fleet is considered the decrease is less than 3% for the less vulnerable species (i.e. 20% encounter mortality) or less than 2% for the vulnerable species (i.e. 80% encounter mortality). At level 6 this varies between a decrease of almost 11% if the ecosystem component mainly occurs in the less fished rectangles to a non-significant increase of 0.3% for the most vulnerable ecosystem component (i.e. an encounter mortality of 80% and mainly occurring in the heavily fished rectangles).

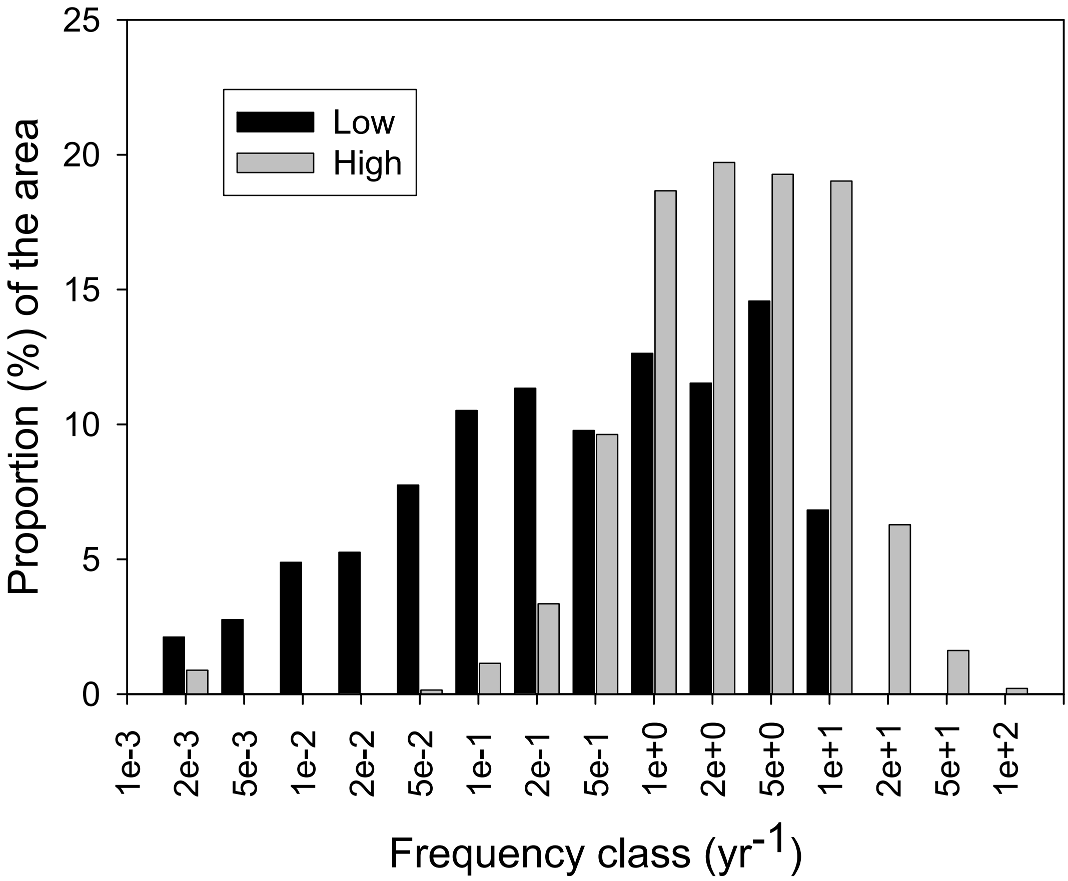


Figure 7. Distribution of trawling frequencies of the fished spatial units at two spatial resolutions: Low (ICES rectangles based on VIRIS) and High (1x1 Nm squares based on APR/VMS).

Discussion

The above example has shown what type of information is required to adequately describe fishing impact and how the pressure indicators at different levels of information content differ in their representation of fishing impact. Higher level indicators provide more accurate time-series that better incorporate changes in the composition of the fleet, fishing practices or modifications of the gear. Moreover, they allow a setting of more realistic reference levels as these pressure indicators can be directly linked to the state indicators which are usually the focus of objectives and policy commitments.

Table 1. Summary of absolute values and trends of the pressure indicators at different levels of information content and for populations that differ in vulnerability to that fishery. Level 1 is the fleet capacity, Level 2 the effort in days-at-sea, Level 3 the frequency (year^{1}) and Level 5 and 6 the population mortality (%). For the spatial distribution of the fleet the distinction is based on the information content of the input, e.g. level 3 (no information) and level 4 at high or low resolution, for level 6 the spatial distribution of the population is based on the distribution in relation to that of the fleet.

Level	Spatial distribution		Encounter Mortality (%)	Value in year 2000	Relative change (%)	R^2	p-value
	Fleet	Population					
1				276	-6.5	0.91	0.00
2				34829	-6.8	0.96	0.00
3				0.51	-2.6	0.64	0.01
5	3	Even	20	6	-7.6	0.79	0.00
5	4 Low	Even	20	9	-2.4	0.63	0.01
5	4 High	Even	20	7	-2.8	0.89	0.00
5	3	Even	80	24	-7.6	0.79	0.00
5	4 Low	Even	80	30	-1.6	0.59	0.01
5	4 High	Even	80	18	-1.4	0.75	0.00
6	3	High	20	6	-7.6	0.79	0.00
6	4 Low	High	20	23	-3.2	0.80	0.00
6	4 High	High	20	12	-1.6	0.75	0.00
6	3	High	80	24	-7.6	0.79	0.00
6	4 Low	High	80	73	-1.9	0.73	0.00
6	4 High	High	80	29	0.3	0.20	0.20
6	3	Low	20	6	-7.6	0.79	0.00
6	4 Low	Low	20	2	-8.6	0.72	0.00
6	4 High	Low	20	2	-10.7	0.80	0.00
6	3	Low	80	24	-7.6	0.79	0.00
6	4 Low	Low	80	6	-8.6	0.72	0.00
6	4 High	Low	80	6	-10.4	0.81	0.00

Figure 8. Spatial distribution of fishing activities of two segments of the Dutch beam trawl fleet expressed by the number of registrations per year based on the period 1994-2003. Upper graph are the eurocutters (enginepower < 300 Hp), lower graph are the large vessels.

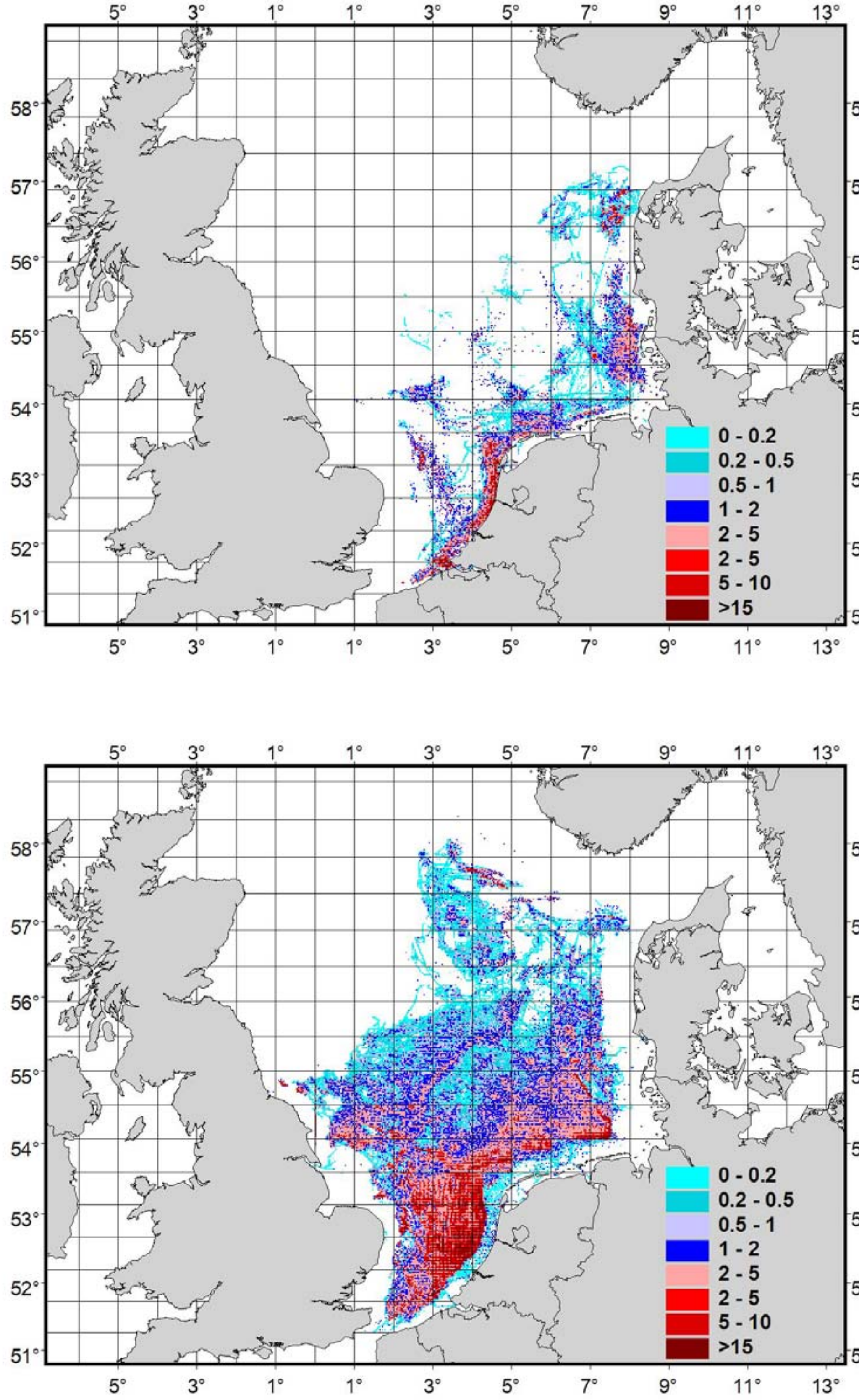
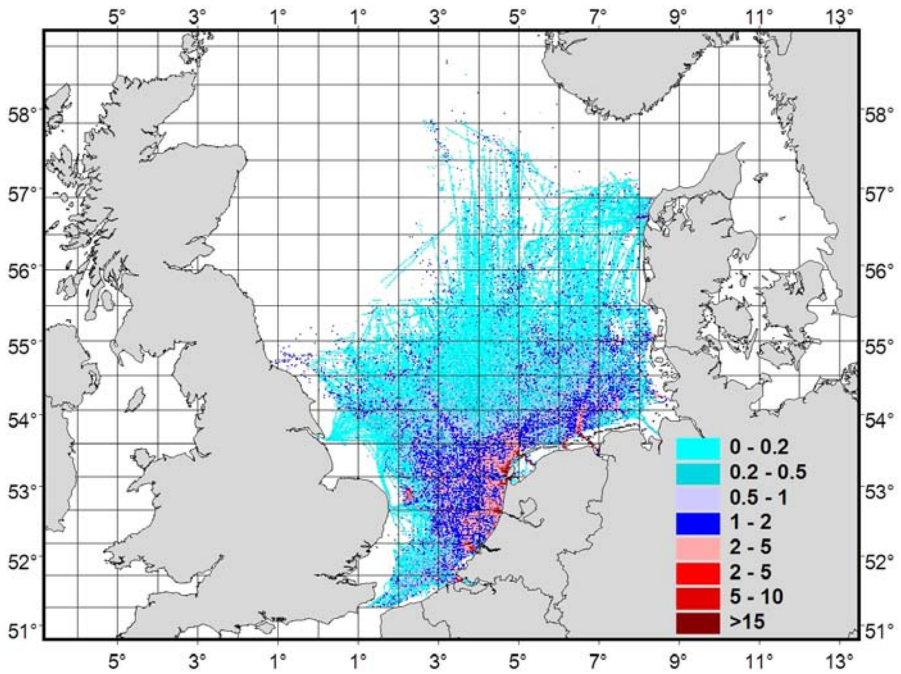
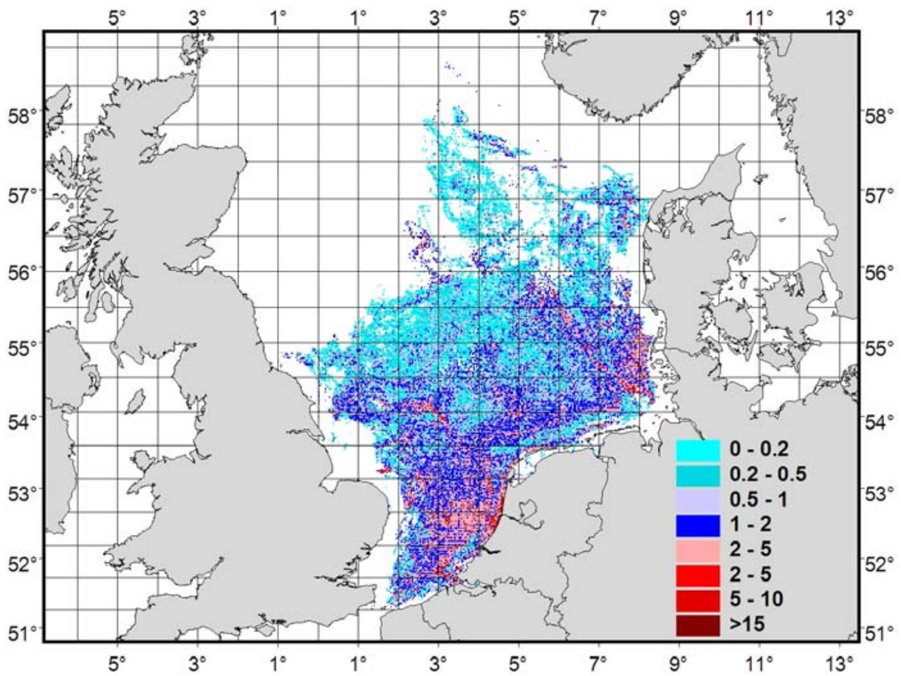


Figure 9. Spatial distribution of activities other than fishing of the Dutch beam trawl fleet expressed by the number of registrations per year based on the period 1994-2003. Upper graph shows positions of vessels steaming (speed > fishing speed), lower graph positions of vessels setting/hauling (speed < fishing speed).





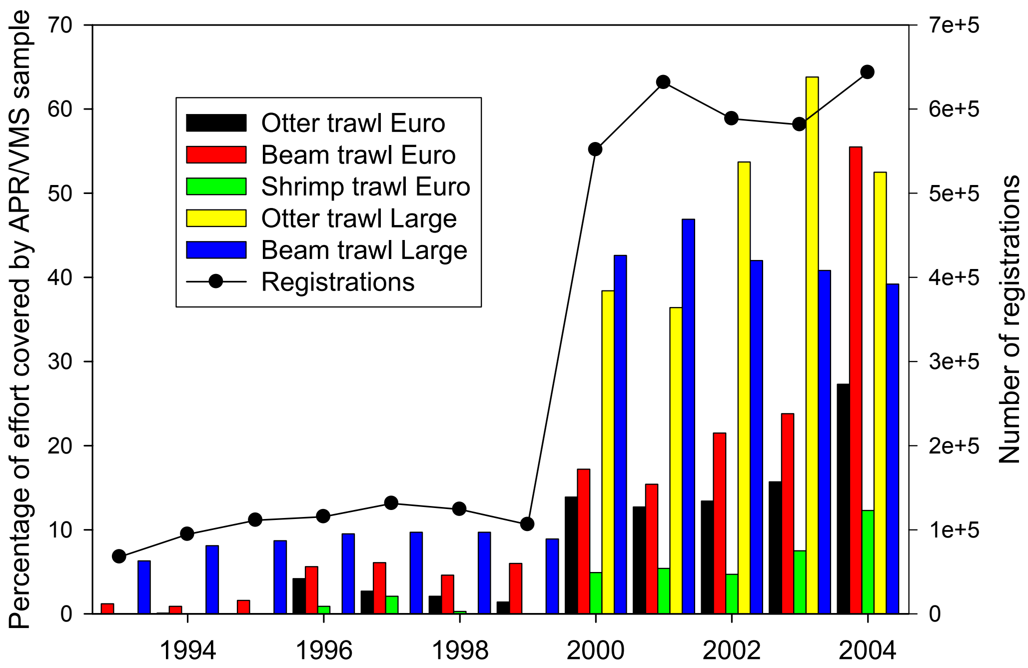


Figure 10. Representativity and composition of the subset of the Dutch beam trawl fleet in the APR/VMS database. Distinguished are subsets of the fleet based on the type of gear (Otter-, Beam- and Shrimp trawl) and engine-power (Large ≥ 300 Hp and Eurocutter). The number of registrations is standardized to a one-hour interval.

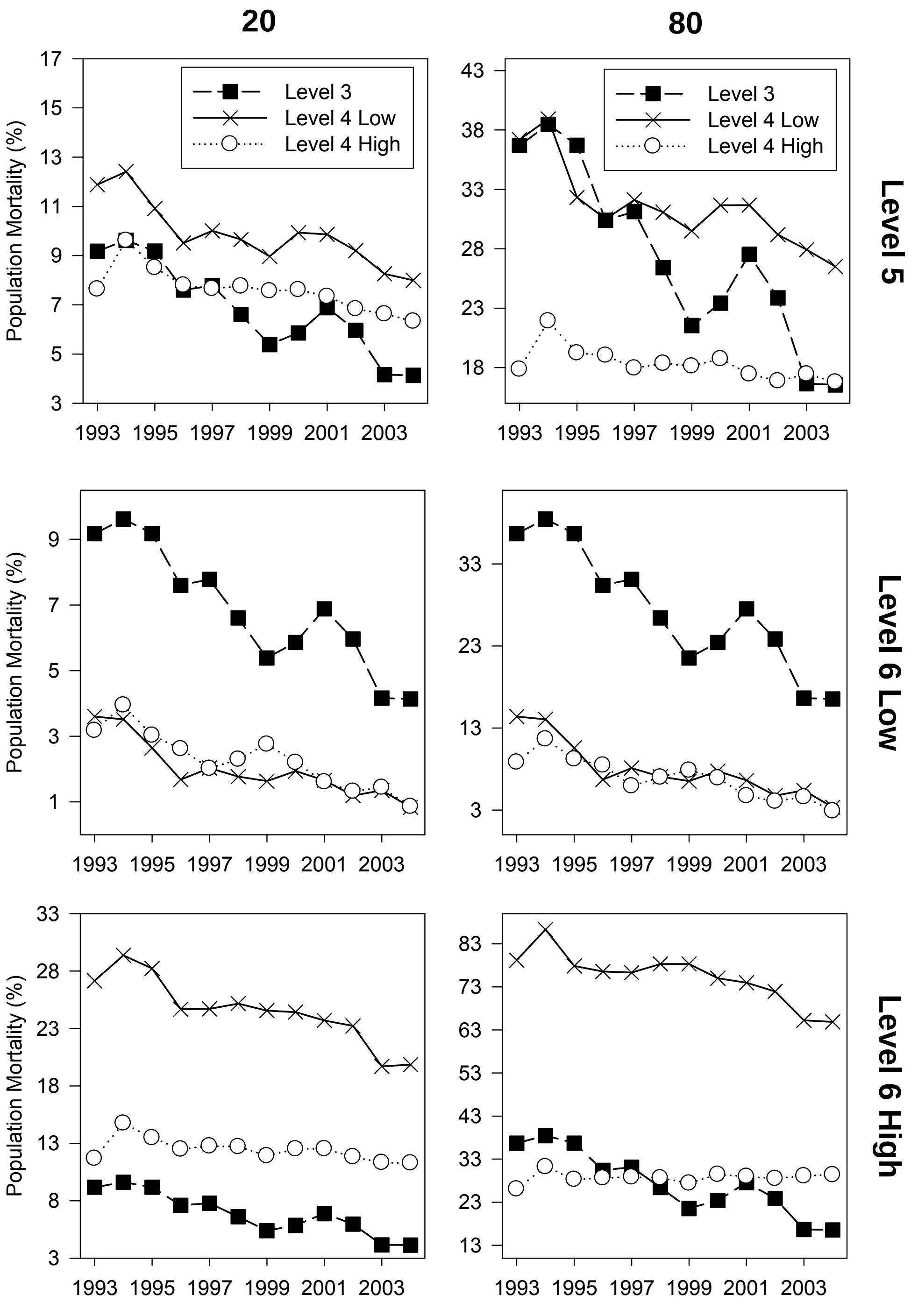


Figure 11. Level 5 and 6 mortality estimates depending for Level 5 (upper) only on encounter mortality (e.g. 20% left, 80% right) for Level 6 on both encounter mortality and the spatial distribution. For Level 6 the estimates "Low" (middle) and "High" (bottom) describe populations that only occur in the ICES rectangles with less respectively more than mean effort, "Even" is a population that is evenly distributed (center). For explanation levels see figure 1.