Potential pressure indicators for fishing, and their data requirements

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

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Indicators of fishing pressure are necessary to support an ecosystem approach to fisheries management (EAFM). We present a framework that distinguishes four levels of pressure indicators that move from being a simple description of anthropogenic activity to more precisely describing the actual pressure on the ecosystem and its components, but which require increasingly more information to be quantified. We use the example of the Dutch beam trawl fleet in the North Sea to compare these pressure indicators, as the level of information used is increased. The first level is that of fleet capacity (e.g. number of vessels), the second is fishing effort, usually expressed as the number of hours fishing or days at sea, the third incorporates fishing parameters such as the proportion of time actually spent fishing, fishing speed, or gear characteristics, e.g. the size of the beam trawl in order to determine the frequency with which an area is fished, and at the fourth level, the most informative measure of fishing pressure, annual fishing mortality, is available for a few commercial species from stock assessments. For other species, it can be calculated from the lower level pressure indicators through the incorporation of the chance of individuals of a species coming into contact with the fishing gear and the encounter mortality, which is the portion of mortality caused by the passing of the gear. Comparison of trends and absolute values shows that the pressure indicators at different levels differ considerably in their description of both present and historical fishing impact in the North Sea. Therefore, for an EAFM, we advise using the highest level pressure indicator that can be obtained with the data available.

Keywords: ecosystem approach to fisheries management, fishing impact, fishing mortality, marine ecosystem, métier, pressure indicator.

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Introduction

When implementing an ecosystem approach to fisheries management (EAFM), indicators are required to describe the pressures affecting the ecosystem, the state of the ecosystem, and the response of managers (Jennings, 2005). Such indicators can be used to support management decision-making, to track progress towards meeting management objectives, and to aid communication with non-specialist audiences (Garcia *et al.*, 2000; Rice, 2000; Rochet and Trenkel, 2003). Many indicators have been proposed (e.g. Rice, 2000; Link, 2002; Link *et al.*, 2002; Rochet and Trenkel, 2003), but few (if any) of those that track changes in the "state" of the marine environment or of different ecosystem components (e.g. fish, benthos, habitat) can support management directly (Rice, 2000). This is largely because the precise causes of any changes in "state" may be poorly understood, making it difficult to specify appropriate management action. To implement an EAFM successfully, therefore, it is not only necessary to have a suite of indicators that accurately and comprehensively portray the "state" of various ecosystem components, but it is also critical to have indicators that describe changes in the level of different manageable anthropogenic activities, and that indicate the impact of each activity on the various ecosystem components. Only by adequately covering both aspects will the mechanistic links between "cause" and "effect" be well enough understood to provide the advice required (Daan, 2005).

Several frameworks have been proposed for classifying environmental management indicators on this basis, e.g. the pressure state response (PSR) system (Garcia and 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. This is in line with traditional fisheries management, in which a report is made for each stock on what is considered the best state indicator (SSB, spawning-stock biomass) and best pressure indicator (*F*, fishing mortality), whereas the response indicator is usually the total allowable catch set by managers. As policy commitments and associated objectives generally relate to "state", for example, to conserve biodiversity, reference points, targets, or indicator trends needed to measure progress towards meeting management objectives tend most frequently to be set for "state" indicators. However, the "state" of different components of marine ecosystems can rarely, if ever, be managed directly. All managers can realistically hope to achieve is to manipulate "pressure" such that "state" indicators are kept within, or moved towards, acceptable limits. "Pressure" and "response" indicators

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Appendix 1
Derivation of optimal observer coverage rates

The optimal sampling fractions can be found using the Cauchy–Schwartz Inequality or Lagrange multipliers. The second component of Equation (2) and the first component of Equation (3) are not a function of f_s, so the minimum of Equation (2) with respect to f_s will not be affected by those components. By the Cauchy–Schwartz Inequality (Cochran, 1977, pp. 96–98; Casella and Berger, 2002, p. 187),

$$(T - T_0)\left[\mathrm{Var}(\hat{\Theta}) - \sum_{s=1}^{S} B_s\right] = \left(T_n \sum_{s=1}^{S} f_s N_s\right)\left(\sum_{s=1}^{S} \frac{A_s}{f_s}\right) \geq \left(\sum_{s=1}^{S} \sqrt{A_s T_n N_s}\right)^2.$$

Given A_s and N_s for all size classes and the cost per trip (T_n), equality is achieved when

$$\chi_s = \left(\frac{A_s/f_s}{T_n f_s N_s}\right)^{1/2} \qquad (11)$$

for each s are equal (i.e. $\chi_s = \chi$). Letting f_s^* denote the sampling fraction for the sth size class that minimizes the variance, we can rewrite Equation (11) as

$$\sqrt{A_s T_n N_s} = \chi T_n N_s f_s^* . \qquad (12)$$

Summing Equation (12) over all S and solving for χ, we can substitute for χ in Equation (11) to obtain

$$f_s^* = \frac{T - T_0}{T_n} \frac{(A_s/N_s)^{1/2}}{\sum_{s=1}^{S} (A_s N_s)^{1/2}}.$$

Recognizing

$$T - T_0 = T_n \sum_{s=1}^{S} f_s N_s = T_n \sum_{s=1}^{S} n_s = T_n n$$

yields the result in Equation (4).

The same approach is used to obtain the optimal sampling fractions with multiple estimation objectives. Noting that the objective function [Equation (7)] can be written as

$$O_2 = \sum_{p=1}^{P} \frac{w_p}{\Theta_p^2}\left[\sum_{s=1}^{S} \frac{A_{ps}}{f_s} + \sum_{s=1}^{S} B_{ps}\right] = \sum_{s=1}^{S}\sum_{p=1}^{P} \frac{w_p A_{ps}}{\Theta_p^2 f_s} + \sum_{s=1}^{S}\sum_{p=1}^{P} \frac{w_p B_{ps}}{\Theta_p^2} = \sum_{s=1}^{S} \frac{O_s}{f_s} + \sum_{s=1}^{S} B_s,$$

we have the same format as Equation (2), and the optimal sampling fractions [Equation (8)] follow.

Appendix 2
Derivation of the unbiasedness of $\hat{A}_\nu$

The estimator $\hat{S}^2_{\Theta,\nu}$ has a random component:

$$\sum_{t=1}^{n_\nu}\left(\hat{\Theta}_t - \hat{\bar{\Theta}}_\nu\right)^2 = \sum_{t=1}^{n_\nu} \hat{\Theta}_t^2 - n_\nu \hat{\bar{\Theta}}_\nu^2 = \frac{n_\nu - 1}{n_\nu} \sum_{t=1}^{n_\nu} \hat{\Theta}_t^2 - \frac{1}{n_\nu} \sum_{\substack{t=1 \\ t \neq u}}^{n_\nu} \sum_{u=1}^{n_\nu} \hat{\Theta}_t \hat{\Theta}_u .$$

The expectations we require are

$$E\left(\sum_{t=1}^{n_\nu} \hat{\Theta}_t^2\right) = E\left\{\sum_{t=1}^{n_\nu}\left[\Theta_t^2 + \mathrm{Var}\left(\hat{\Theta}_t\right)\right]\right\} = \frac{n_\nu}{N_\nu} \sum_{t=1}^{N_\nu}\left[\Theta_t^2 + \mathrm{Var}\left(\hat{\Theta}_t\right)\right]$$

and

$$E\left(\sum_{\substack{t=1 \\ t \neq u}}^{n_\nu} \sum_{u=1}^{n_\nu} \hat{\Theta}_t \hat{\Theta}_u\right) = E\left(\sum_{\substack{t=1 \\ t \neq u}}^{n_\nu} \sum_{u=1}^{n_\nu} \Theta_t \Theta_u\right) = \frac{n_\nu(n_\nu - 1)}{N_\nu(N_\nu - 1)} \sum_{\substack{t=1 \\ t \neq u}}^{N_\nu} \sum_{u=1}^{N_\nu} \Theta_t \Theta_u,$$

which rely on the facts that $\mathrm{Var}(x) = E(x^2) - E(x)^2$, and the estimators $\hat{\Theta}_t$ are independent across elements in the stratum. We also rely on the first and second-order inclusion probabilities in SRS, n_ν/N_ν and $n_\nu(n_\nu - 1)/N_\nu(N_\nu - 1)$, respectively, which are proven elsewhere (Särndal *et al.*, 1992, pp. 31–32). Now,

$$E(\hat{S}^2_{\Theta,\nu}) = \frac{1}{n_\nu - 1} E\left[\sum_{t=1}^{n_\nu}\left(\hat{\Theta}_t - \hat{\bar{\Theta}}_\nu\right)^2\right] = \frac{1}{N_\nu} \sum_{t=1}^{N_\nu} \Theta_t^2 - \frac{1}{N_\nu(N_\nu - 1)} \sum_{\substack{t=1 \\ t \neq u}}^{N_\nu} \sum_{u=1}^{N_\nu} \Theta_t \Theta_u + \frac{1}{N_\nu} \sum_{t=1}^{N_\nu} \mathrm{Var}(\hat{\Theta}_t) = S^2_{\Theta,\nu} + \frac{1}{N_\nu} \sum_{t=1}^{N_\nu} \mathrm{Var}(\hat{\Theta}_t),$$

which gives unbiasedness of $\hat{A}_\nu$ (i.e. $E(\hat{A}_\nu) = N_\nu E(\hat{S}^2_{\Theta,\nu}) = A_\nu$).

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are clearly essential in this process. Such indicators also often have the desirable properties of ease of measurement and rapid response times. Consequently, 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 and Jennings, 2004).

The development of pressure indicators for fisheries has in the past tended to be hampered by confusion over the difference between the actual ecological impact of fishing (mortality and habitat change) and the community level changes that are later seen as a consequence of this impact (e.g. a change in the size structure of the community). The ecosystem components for which most information is available on the direct effects of fishing are fish and benthic invertebrates (Dayton *et al.*, 1995; Jennings and Kaiser, 1998; Hall, 1999; Collie *et al.*, 2000; Kaiser and de Groot, 2000). Many indicators have been proposed that describe the state of these components at different hierarchical levels (e.g. population and community levels, see Frid *et al.*, 2005; Piet and Jennings, 2005). However, the state of the individual components is not only determined by the effects of a particular anthropogenic activity such as fishing, but by a combination of all intrinsic and extrinsic factors that combine to structure populations and communities. These include 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 and Frid, 2001; Kröncke, 2001). In theoretical ecology terms, impact or disturbance is the mortality caused by perturbations to the ecosystem, and fisheries impact is an anthropogenic

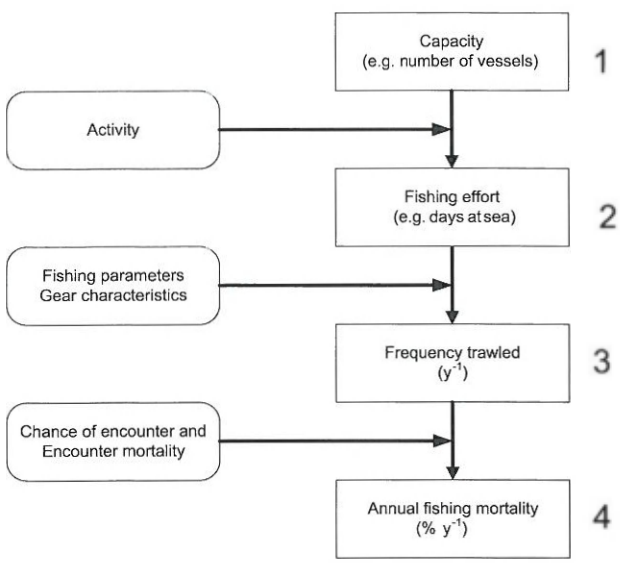

Figure 1. Schematic representation of fishing pressure indicators at different levels of information content. Activity indicates the number of fishing hours or days at sea per vessel. The fishing parameters and gear characteristics determine how much area is covered by a unit of fishing effort, which translates into the frequency with which a specific area is trawled. The chance of encounter determined by the spatial distribution of the population relative to that of the fishery gear and the encounter mortality, expressed as the proportional mortality of individuals in the path of the gear, determine the extent to which a population is actually affected by the fishery (i.e. the annual fishing mortality rate).

source of mortality. Clearly, to be able to predict realistically the response of ecosystem components to fisheries impact, 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.

In terms of indicators required to implement an EAFM, pressure indicators should ideally account for mortality to an ecosystem component that results directly from fishing, state indicators for the overall state of that component, and response indicators for the response of managers required to alter the level of pressure where there have been unacceptable changes in state. In case management objectives are set for populations of ecosystem components other than the target fish species (e.g. threatened and declining species, sensitive benthic species) or at the community level (e.g. mean weight of the fish community), the known mortality estimates of a few target fish species that come from stock assessments may not be adequate to link pressure with such state indicators. For those cases, the mortality of all ecosystem components can be calculated following the swept-area method introduced simultaneously by Pope *et al.* (2000) for fish and Piet *et al.* (2000) for benthos.

Starting from the premise that fishing mortality is the most accurate measure to describe fishing impact but acknowledging that data limitations often force us to use less-informative proxies, we introduce a framework (Figure 1) that encompasses the most common pressure indicators. The main objective of this paper is to describe this framework and how the different existing pressure indicators compare with one another, by incrementally adding information to the most basic pressure indicator, ultimately leading to indicators that describe the actual ecological impact of fishing, i.e. the level of mortality inflicted on a particular ecosystem component. This approach makes explicit the assumptions that are made at lower levels of information content and how these influence the pressure indicator.

We chose the Dutch beam trawl fishery in the southern North Sea as a case study. Beam trawling accounts for a high proportion of all fishing activity, particularly in the southern North Sea (Jennings *et al.*, 1999), and the Dutch beam trawl fleet is responsible for >70% of total beam trawl effort in the North Sea. Moreover, the fishery and fleet have been intensively studied in recent years, resulting in a high level of knowledge regarding its precise operation (Rijnsdorp *et al.*, 1998, 2000a, 2000b; Piet *et al.*, 2000). Mortality is only estimated for the two target species of the beam trawl fishery, plaice (*Pleuronectes platessa*) and sole (*Solea vulgaris*), despite the fishery's potential to cause collateral damage to other components of the marine ecosystem. For estimates of fishing pressure on these other components, only one of the least informative measures of effort (i.e. days at sea) at a spatial scale of ICES rectangles (~30 × 30 nautical miles) is available. In order to exemplify how this can be improved, we used the swept-area method to estimate the mortality of a "virtual population", with specific characteristics that determine the impact the fishery has on it.

The consequences of these findings for the collection of data needed to support an EAFM are discussed.

Material and methods

The Dutch beam trawl fleet operates mainly in the southeastern North Sea and targets the flatfish species plaice and sole. Two principal fishing métiers are usually distinguished: "large vessels" with an engine power of 221 kW or more, and "eurocutters", with an engine power <221 kW. The two métiers differ markedly in fishing practice and gear characteristics. Typically, the large vessels deploy 2 × 12-m beam trawls and are not allowed to fish inside the 12 mile coastal zone or the "plaice box", whereas the eurocutters deploy 2 × 4-m beam trawls and are allowed to fish inside those areas.

Because beam trawling has such a high potential to cause collateral damage to other components of marine ecosystems, including fish and benthic invertebrate communities and seabed habitat, it has long been the focus of considerable scientific attention. This has led to the collection of a wide range of data, including information on the capacity, spatial coverage, and behaviour of different types of vessels within the fleet, and temporal and spatial variation in these at different scales (Rijnsdorp *et al.*, 1998; Piet *et al.*, 2000). Two databases that differ in their spatial resolution were analysed in this study.

(1) *Low spatial resolution*. The VIRIS database contains information on fishing activities of the entire Dutch fleet at a spatial resolution of ICES rectangles stored in individual fishers' EC logbooks. Data were extracted for the years 1994–2004. The database distinguishes different segments of the fleet based on their engine power, and contains information on the time and date of the start and end of the fishing trip, the gear used, the ICES rectangles fished, and the landings by fish species. It is designed for quota management, but is available for research purposes, and similar databases are available for other EC countries.

(2) *High spatial resolution*. The APR/VMS database consists of automated position registration (APR) and vessel monitoring through satellite (VMS) data at a resolution of 1 min latitude × 2 min longitude (~1×1 nautical mile) spatial units, which are referred to subsequently in text as squares. APR data are based on a sample of about 10% of the Dutch beam trawl fleet that was equipped with APR equipment for the period 1993–2000, during which the position of the vessels was recorded every 6 min (Rijnsdorp *et al.*, 1998). The VMS data became available from 2000, when positions of all EU vessels >24 m were recorded for enforcement purposes. From September 2003, this was extended to vessels >18 m, and subsequently from the 1 January 2005 to vessels >15 m. Positions are recorded approximately every 2 h. Although the data are collected by all EC countries for enforcement purposes, not all countries have access to VMS data for research purposes. For the Dutch beam trawl fleet, VMS data from only a subset of ~40% of the vessels that allowed the use of their VMS registrations are available for research purposes. In addition to detailed data on track positions, some vessels provided data on a haul-by-haul (HBH) basis of the catch of the target species, trawling speed, and times of shooting and hauling of the gear.

The ecological impact of fishing by the Dutch beam trawl fleet can be described by pressure indicators at four levels of increasing information content (Figure 1). Level 1 quantifies fleet capacity, i.e. the number of vessels in a fishery, where different fishing métiers (defined by target species, fishing gear used, and area visited; Laurec *et al.*, 1991) may be defined as necessary. Level 2 is the measure most commonly referred to as fishing effort, calculated as fleet capacity (usually in number of vessels, but it may also take account of vessel tonnage or engine power) multiplied by their activity (e.g. number of hours fishing or days at sea). At Level 3, pressure is described by the trawling frequency and includes information on fishing practice and gear characteristics, allowing, for example, the total area of seabed swept by the gear, or the volume of water filtered, in a given period of time to be calculated. At this level, it becomes relevant if information on the spatial distribution of effort exists and when this information is available, at what spatial resolution. We evaluated this by distinguishing between: (i) no spatial information available; (ii) low spatial resolution; or (iii) high spatial resolution. Finally, at Level 4, we have the ultimate measure of fishing impact: annual fishing mortality. For the target species of the beam trawl fishery, plaice and sole, this is available from stock assessments. Alternatively, it can be calculated following the swept area method (Piet *et al.*, 2000; Pope *et al.*, 2000) for all ecosystem components, from the lower level pressure indicators through the incorporation of information on the species-specific mortality rate per individual contact with the gear, combined with information on the overlap in spatial distribution of populations of different organisms of concern with fishing activity.

Level 1: Fleet capacity

The number of registered Dutch beam trawl vessels belonging to each of the two métiers based on engine-power was determined for each year from the VIRIS database.

Level 2: Fishing effort

Total annual fishing effort, in terms of the number of days at sea, was determined for each métier within the Dutch beam trawl fleet based on the VIRIS database.

Level 3: Frequency trawled

The total area of seabed swept by the Dutch beam trawl fleet in any given year (swept area, SA, in $m^2\ y^{-1}$) was estimated from

$$SA = E * HF * S * 1852 * 2\,W \qquad (1)$$

where E is the measure of effort, i.e. the number of days recorded at sea by the entire fleet ($d\ y^{-1}$), HF the mean number of hours fished in a day ($h\ d^{-1}$), S the mean trawling speed (knots, converted to $m\ h^{-1}$ by multiplying by 1852), and W is the width of each beam (m) with two beam trawls towed by each vessel. These parameter values, determined using information held in both databases, varies between the different métiers in the fleet, so SA needs to be calculated for each métier independently. Summing the estimates of SA for each métier produced the estimate of SA for the entire Dutch beam trawl fleet. In addition to the EC logbook data, trawling speed was recorded by a sample of fishing vessels in the HBH data, allowing the determination of the range of possible fishing speeds. Vessel speed could also be calculated from the APR data by calculating the distance between subsequent positions and dividing by 0.1 h (i.e. 6-min time intervals). The VMS database provided the measured speed at each position for most records. The number of hours spent fishing in each day (HF) required fishing activity to be distinguished from other vessel activity, i.e. steaming, shooting, or hauling the gear, or drifting. This distinction was made on the basis of vessel speed where, for each métier in the fleet, the mean proportion of records per 24 h period that lay within the fishing speed range was interpreted as the proportion of the time spent fishing. The mean time per day spent fishing was calculated from the HBH data that contained the times of shooting and hauling of the gear.

The APR/VMS database holds registration data recorded at different time intervals. In order to combine these data sets, the total number of hours fishing per year was calculated for each set as follows:

$$E * \mathrm{HF} = \frac{\mathrm{FR} * \mathrm{TI}}{\mathrm{PD}} \tag{2}$$

where FR is the number of fishing registrations, time interval (TI) is equal to 0.1 h (6 min) for APR and ~2 h for VMS, and PD is the proportion of the fleet in the APR or VMS sample (i.e. for which data are recorded in the database). Note that the right side of this equation can be substituted directly into Equation (1). The mean trawling frequency (TF) within the area of Dutch beam trawling operations was calculated as

$$\mathrm{TF} = \frac{\sum_{i=1}^{\mathrm{SU}} (\mathrm{SA}_i/A_i)}{\mathrm{SU}}, \tag{3}$$

where SA_i is the area of seabed swept by the Dutch beam trawl fleet in spatial unit i with total area A_i. The area A_i was calculated using GIS (projection UTM−1984, zone 31 N). Whether or not spatial information on fishing activities was available, and if so at what resolution, determined the number of spatial units (SU). If based on the VIRIS database or APR/VMS database, SU equalled, respectively, the number of ICES statistical rectangles or 1 × 2 min squares in which Dutch beam trawl activity had been recorded for that year. The North Sea was defined as ICES area IV, minus the area deeper than 200 m, and if no spatial information was available, SU=1 and A_1 equalled the area of the North Sea as defined.

Level 4: Annual fishing mortality (%)

Ideally, the effect of a fishery on any ecosystem component should be expressed as an annual fishing mortality (i.e. the percentage fishing-induced number of deaths per year in relation to population abundance). In traditional fisheries science, the instantaneous fishing mortality rate F is usually calculated as part of the stock assessment process, and it can be converted easily to the more readily understood concept of annual fishing mortality through

$$\mathrm{AFM} = 100 * (1 - \exp^{-F}). \tag{4}$$

F can be expressed as catchability × effort, where catchability refers to the chance that an individual in the population is killed by the gear (Beverton and Holt, 1957). This depends on (i) the chances of individuals of a species coming into contact with the fishing gear, which is determined by the distribution of the species in relation to the distribution of the fleet, and (ii) gear efficiency, which is the proportion of the population in the path of the gear that is retained by the gear (Ricker, 1975). Catchability therefore integrates all aspects of the distribution of the population in relation to that of the fishing fleet, crew skills, vessel characteristics, and gear efficiency (Rijnsdorp *et al.*, 2006).

Fishing mortality estimates of plaice and sole were available from the ICES stock assessments. For plaice, F estimates are based on both landings and discards, for sole on landings only. As the stock assessments for these species are known to have underestimated mortality in the past few years, the values for the final year were not presented.

For all non-target species for which mortality estimates cannot be obtained from the stock assessments, we use the swept-area method (Piet *et al.*, 2000; Pope *et al.*, 2000) and calculated AFM (%) summed over all spatial units as

$$\mathrm{AFM} = \frac{100 * \sum_{i=1}^{\mathrm{SU}} N_i\{1 - (1 - \mathrm{EM})^{\mathrm{TF}_i}\}}{\sum_{i=1}^{\mathrm{SU}} N_i}, \tag{5}$$

where N_i is the population abundance and TF_i the trawling frequency in spatial unit i, and EM is the encounter mortality, which differs from gear efficiency because it encapsulates all forms of mortality caused by the fishing gear, including those that do not result in retention of the animal (Bergman and van Santbrink, 2000). Encounter mortality depends on characteristics of the species (e.g. size, position in the water column or sediment, fragility, and swimming speed) in relation to gear characteristics (e.g. type of gear, mesh size; Wardle, 1988; Casey, 1996; Collie *et al.*, 2000). Similar to Level 3, the resolution at which spatial information is available is relevant, but at Level 4 this applies both to the fishery and the species concerned. Moreover, the resolution of the fishery information determines the required resolution of the species information for optimal estimation of the annual fishing mortality. Therefore, there is no point in acquiring spatial information of the species at a resolution beyond that available for the fishery.

In order to illustrate how encounter mortality and overlap in distribution of the species with the fishery determine the value of the indicator, we distinguish virtual populations that differ in these two characteristics. Published gear efficiency rates of benthic invertebrates lie within the range 0–90% (Bergman and van Santbrink, 2000; Collie *et al.*, 2000). In our example, therefore, we used encounter mortality rates of 20 and 80%. For the overlap in distribution, we distinguished a species with "high overlap" (HO), which is aggregated in areas where fishing activity is highest (i.e. in spatial units where TF_i is greater than the mean TF across all spatial units), and a species with "low overlap" (LO), which is aggregated in areas where fishing activity is least ($\mathrm{TF}_i <$ mean TF). In all cases, the population with a fixed number of organisms was uniformly distributed over the spatial units that matched the criteria. More spatial units result in less organisms per unit.

We explored the same scenarios as for the Level 3 indicator: no spatial information, spatial information at low resolution (ICES rectangles), and spatial information at high resolution (1 × 2 min squares). For comparison of indicators across levels, we fitted linear regressions to the centred time-series (centred at year 2000), and used the slopes and intercepts to calculate the estimated value in year 2000 (intercept), and the relative change (%) over time (slope × 100/intercept).

Results

Level 1: Fleet capacity

According to the VIRIS database, the total number of registered beam trawlers declined from 378 in 1995 to 224 in 2004 (Figure 2). The reduction in the number of large vessels was much smaller in relative terms than the reduction in the number of eurocutters, so the proportion of large vessels within the fleet increased from 55% in 1995 to 63% in 2004 (Figure 2).

Level 2: Fishing effort

On the basis of the VIRIS database, the activity per vessel (days at sea per year) varied considerably within and between métiers

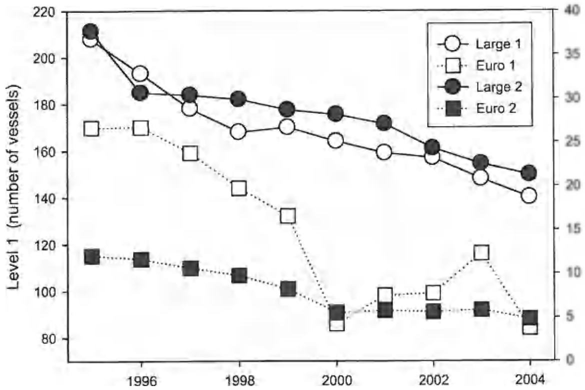

Figure 2. Time-series of the pressure indicator at Level 1 (fleet capacity, open symbols) and Level 2 (fishing effort, shaded symbols) for two métiers within the Dutch beam trawl fleet. (Large 1 and Large 2 are, respectively, the Levels 1 and 2 indicators for "Large vessels"; Euro 1 and Euro 2 are, respectively, the Levels 1 and 2 indicators for "Eurocutters").

(Figure 3). Overall, 87% of the large vessels spent 150–250 d at sea per year, with an average of 170 d at sea per year. For eurocutters, mean activity was much less, with an average of just 67 d at sea per year, but the distribution was skewed because 25% of eurocutters registered fewer than 10 d at sea per year, many registering only 1 d at sea per year. For both métiers, activity per vessel decreased by about 1.5 d at sea per year. Total Dutch beam trawl effort decreased from 49 765 d at sea in 1995 to 26 034 d at sea in 2004. Over the same period, the proportion of total Dutch beam trawl fishing effort undertaken by large vessels increased from 76 to 82% (Figure 2).

Level 3: Frequency trawled

For each of the two fishing metiers, frequency distributions of the estimated vessel speed for all APR and VMS registrations were compared with frequency distributions of recorded trawl speeds in the HBH data (Figure 4). Fishing speeds in the HBH data ranged from 3 to 6 knots for eurocutters, and from 5 to 8 knots for large vessels. APR and VMS records giving speed estimates within these ranges were considered to be "fishing" records, and the proportion

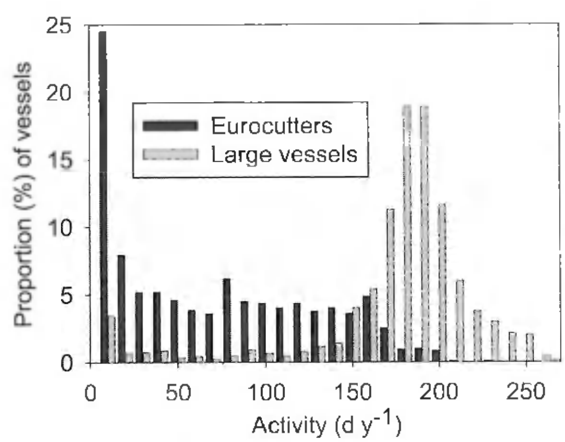

Figure 3. Vessel activity frequency distributions determined over the period 1995–2004 for each métier of fishing vessel within the Dutch beam trawl fleet.

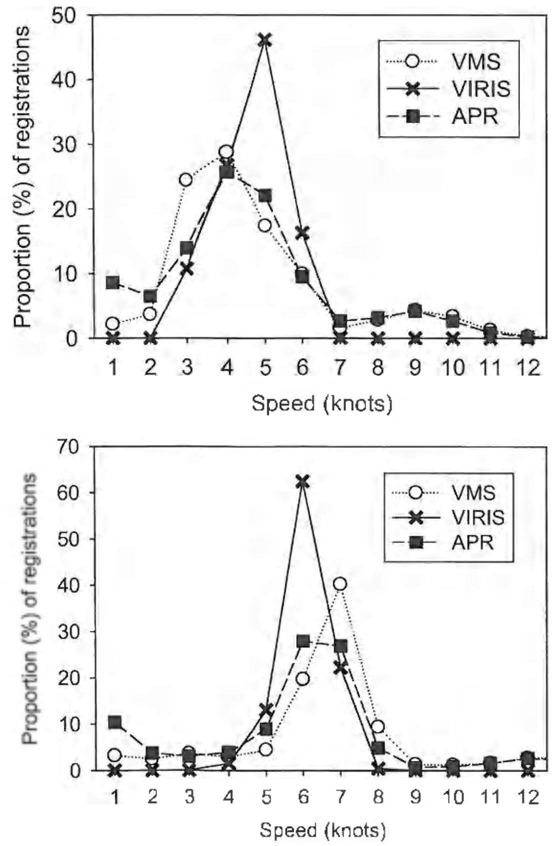

Figure 4. Estimates of trawling speed derived from each of the three database sources for "eurocutter" (upper panel) and "Large vessel" (lower panel) métiers within the Dutch beam trawl fleet.

of records falling into this category provided an estimate of the proportion of time spent fishing, allowing the number of hours spent fishing by each métier in an average 24 h period to be calculated (Table 1). There was little difference between the estimated fishing speeds or the proportion of time spent fishing derived from either the APR or the VMS databases (Table 1), so the values derived from the VMS database were used in Equation (1) to estimate the area of seabed swept per day by a fishing vessel belonging to each métier. Eurocutters swept an area of 1.2 km^2 on average each day, and large vessels swept an area of 5.3 km^2.

The VIRIS database provided information that identified all the ICES rectangles in which Dutch beam trawlers were recorded fishing in each year. Likewise, the APR/VMS database identified the fished rectangles. Knowing the area of each rectangle or square, and summing over all rectangles or squares in which fishing occurred, allowed the proportion of the total area fished annually by Dutch beam trawling operations to be estimated, depending on the spatial resolution of the data (Figure 5a). The low-resolution data indicated that just over 100 ICES rectangles, amounting to about 58% of the North Sea, was fished annually at the start of the time-series, declining to about 50% annually at the end. In contrast, the high resolution data indicated that considerably less of the North Sea was fished annually, respectively 20% (about 26 000 rectangles) and 14%. Figure 5b shows the time-series of the Level 3 indicator, the frequency fished (TF). The difference in frequency distribution between the VIRIS and APR/VMS data sets is reflected in Figure 6, which shows the occurrence

Table 1. Fishing parameters of two métiers of the Dutch beam trawl fleet in the North Sea based on different data sources.

Métier	Data source	Speed (knots)	Hours fishing per 24 h	Proportion of the day spent fishing (%)	Area (km^2) swept per day
Eurocutters	HBH	4.7	19.4	80.9	1.3
	APR	4.4	17.1	71.1	1.1
	VMS	4.2	19.3	80.4	1.2
Large vessels	HBH	6.1	20.4	85.1	5.5
	APR	6.1	16.3	67.8	4.4
	VMS	6.7	17.7	73.9	5.3

of spatial units with specific trawling frequencies (per year). Frequencies above 20 per year were only observed for the high-resolution APR/VMS data. The two métiers are strongly segregated spatially, because the eurocutters, which are allowed to fish anywhere, appear to avoid the areas where the large vessels fish and concentrate in the 12-mile zone and the plaice box, where the large vessels are not permitted (Figure 7).

The degree to which the subset of the Dutch bottom trawling fleet for which APR/VMS data were 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 8). In the first period (before 2000), mainly large beam trawlers (15–24) and a few eurocutters (1–6) per year were included in the sample. From 2000 on, this increased to 66–143 large vessels and 17–37 eurocutters per year.

Level 4: Annual fishing mortality

The fishing mortality (F) of plaice and sole is estimated as part of the stock assessment process (ICES, 2005) and translated into the annual fishing mortality using Equation (4) (Figure 9). The time-series of this indicator reveals that contrary to what was observed for the lower level indicators, there was no decrease in annual fishing mortality over the 10 y 1995–2004, but rather a slight (but not significant) increase of 0.04% for sole and 0.55% for plaice (Table 2).

For non-target species, there are no stock assessments, so annual fishing mortalities were estimated following the swept-area method. This resulted in time-series of annual fishing mortality for different virtual populations, with various scenarios depending on the spatial resolution of the fishing data and how the spatial resolution of the virtual population being impacted related to that of the fishery (Figure 10). The time-series of annual fishing mortality differed considerably between scenarios, both in terms of absolute value and the relative change (Figure 10, Table 2).

Overall, annual fishing mortality increased with increasing overlap of the population with the fishery, encounter mortality, and spatial resolution, and in the example varies between 0.7 and 80.1%. The relative change in annual fishing mortality over time also varied between scenarios, from a 1.8% decrease to a 1.8% increase.

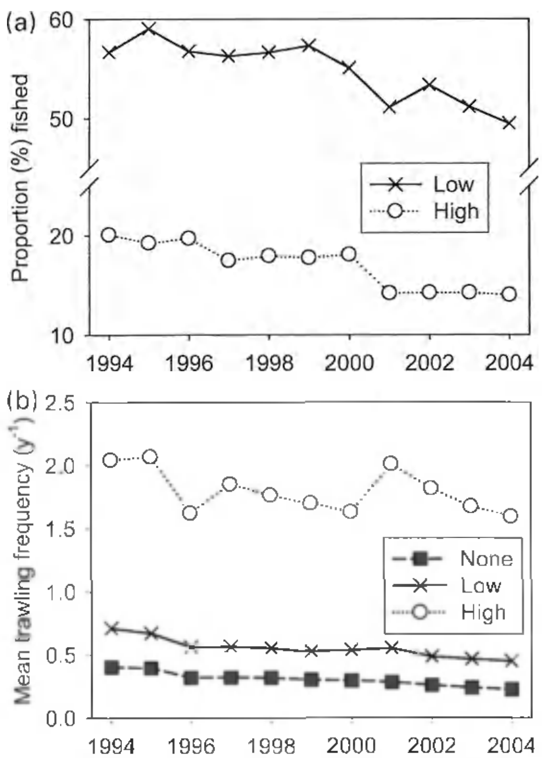

Figure 5. Time-series of the pressure indicator at Level 3. The figures show (a) the variation in the proportion of the total area of Dutch beam trawl operations fished in each year and (b) the mean frequency that the "fishable" area was trawled each year, depending on whether or not information on the spatial distribution is available and at what resolution: high (ICES statistical rectangles) or low (1 × 2 min squares). If no spatial information was available, the proportion of the area fished is by definition 100%.

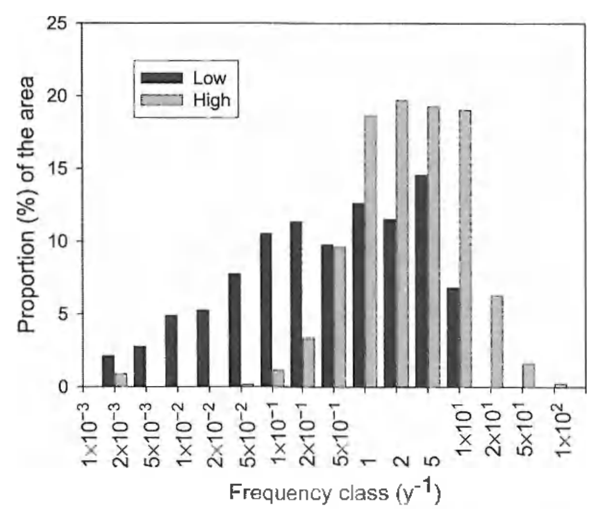

Figure 6. The distribution of fishing frequencies of the individual fished spatial units across the whole area of Dutch beam trawl operations at two spatial resolutions: low (ICES rectangles based on VIRIS) and high (1 × 2 min squares based on APR/VMS).

Table 2. Summary of absolute values and relative trends of the pressure indicators at different levels of information content, and for populations that differ in vulnerability to that fishery.

Level	Distribution population	Encounter mortality (%)	Spatial resolution	Value in 2000	Relative change (%)
1				276	−6.5
2				34 829	−6.8
3			NS	0.3	−5.7
			Low	0.4	−4.1
			High	1.8	−1.6
4		Plaice		50.9	0.5
		Sole		43.9	0.0
	Low overlap	20	NS	0.7	−0.2
			Low	5.1	−0.2
			High	14.2	0.5
		80	NS	2.8	−0.6
			Low	15.4	−0.4
			High	47.2	1.8
	Even distribution	20	NS	0.7	−0.2
			Low	5.5	−0.4
			High	26.1	0.2
		80	NS	2.8	−0.6
			Low	18.4	−1.1
			High	64.2	1.4
	High overlap	20	NS	0.7	−0.2
			Low	5.9	−0.5
			High	37.2	−0.8
		80	NS	2.8	−0.6
			Low	20.7	−1.8
			High	80.1	0.1

Level 1 is the fleet capacity, Level 2 the effort in days at sea, Level 3 the frequency trawled (per year), and Level 4 the annual fishing mortality (%). For the spatial distribution of the fleet, the distinction is based on the information content of the input, e.g. no information, or spatial information at high or low resolution. For the spatial distribution of the population, the distinction is based on the distribution in relation to that of the fleet.

calculated speed according to the APR data is underestimated because (i) the vessel does not follow a straight line between two subsequent registrations, and (ii) if the vessel starts hauling between two registrations, the calculated mean speed will decrease. These points will become increasingly important and result in an underestimation of the speed as the time interval between registrations is increased (as in the VMS data). Therefore, in order to use speed to distinguish fishing registrations from other activities that do not impact the ecosystem, it is necessary to obtain information on the activity of a vessel (e.g. HBH), and to combine this with real speed measurements (i.e. not calculated from intervals between registrations). If this needs to be based on calculated speed values, the interval needs to be set as short as possible. This approach to use speed to distinguish fishing from other activities will never be absolutely perfect, because vessels may engage in activities other than fishing at speeds within the fishing speed range, resulting in spurious fishing position registrations and an overestimation of impact.

If available, information on the spatial distribution of the fishery at the highest possible resolution needs to be incorporated when assessing the pressure on the ecosystem. We used two sources of data that differed in their spatial resolution: VIRIS data are at a relatively low resolution of ICES rectangle ($\sim$30 × 30 nautical miles), whereas the APR/VMS data were aggregated at a relatively high resolution of 1 × 2 min squares ($\sim$1×1 nautical mile). Comparison of the frequency distributions of these data sets shows that the resolution used makes a big difference. According to low resolution data, >50% of the North Sea is fished, of which one-third of the ICES rectangles are fished more than once a year and maximum frequency is about five times per year. According to the high resolution data, only some 20% of the North Sea is fished, of which two-thirds are fished more than once a year and the maximum frequency is up to 100 times per year. In this example, the proportion of the area fished according to the high resolution data may be slightly underestimated, because not all beam trawlers are part of the APR/VMS database available for scientific purposes. The calculated frequency was not affected by this, though, because it was weighted by a factor that raised the frequency of the sample to that of the entire fleet. The important take-home message here is that as the spatial resolution of fishing activity data increases, the proportion of the area fished declines, whereas the trawling frequency within the fished area increases on average.

Level 4: Annual fishing mortality

The annual fishing mortality differs from the lower level indicators, because it explicitly incorporates both the chance of individuals of a species coming into contact with the fishing gear and the encounter mortality, and also because it is the only indicator that describes the pressure relative to the abundance of the

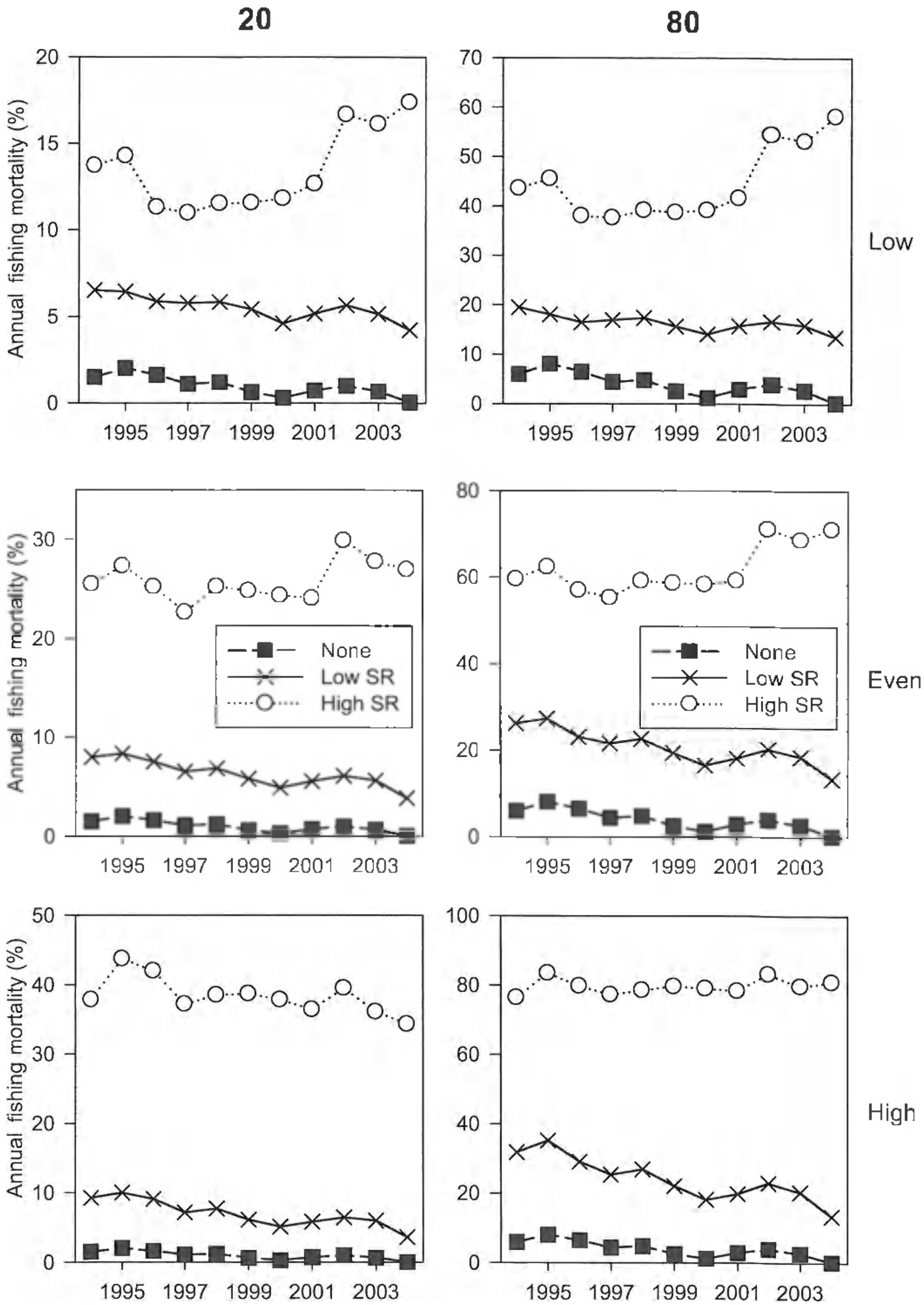

Figure 10. Time-series of the pressure indicator at Level 4 expressed as annual fishing mortality (%) depending on the encounter mortality, i.e. 20 (left) vs. 80% (right), and the distribution of the population in relation to the fishery, i.e. low overlap (top), even distribution (middle), and high overlap (bottom). In each plot, the time-series were calculated without spatial information on the fishery (None), or a spatial resolution of ICES squares (Low SR), or a spatial resolution of 1 × 2 min squares (High SR).

species. As such, it should formally be considered an impact indicator, but the distinction between pressure and impact indicators is not made in the PSR framework.

The time-series of annual fishing mortality based on stock assessments of plaice and sole show that pressure on these stocks in the 10 y 1995–2004 probably increased. This becomes even more likely when considering the systematic underestimation of mortality in the past few years by the stock assessment process (Piet and Rice, 2004). The parameters that describe the annual population mortalities of plaice and sole based on the stock assessments fall within the range observed for the different populations at high spatial resolution based on the modelled pressure indicators, suggesting that a higher spatial resolution provides more accurate estimates.

Although the increase in population mortality of plaice and sole is systematically underestimated in their stock assessments, the increase in population mortality based on the swept-area method is probably also underestimated, because we assumed encounter mortality to be stable over time, failing to take account of technical creep and increases in skipper skills which, according to Rijnsdorp *et al.* (2006), increase the partial fishing mortality rate in the beam trawl fishery (>221 kW) by 2.8% per year for sole and 1.6% for plaice. To what extent these values apply to non-target species, however, remains to be assessed.

When using the swept-area method to calculate the annual fishing mortality, it may theoretically vary between 0 and 100% depending on encounter mortality of that particular species–métier combination, the spatial distribution of the population relative to that of the fleet, and the resolution of the spatial distribution data. In this study, the annual fishing mortality increases with increasing encounter mortality (i.e. 20 → 80%), increasing overlap (i.e. low → even → high), and increasing spatial resolution (i.e. ns → low → high). The increase of mortality with spatial resolution is because, in the calculations, the population is distributed relative to the mean trawling frequency (TF) which, as Figure 5 shows, depends on the spatial resolution. The choice of a fixed TF, independent of the spatial resolution, would probably result in an inverse relationship with spatial resolution, because at a higher resolution, less of the area is fished, although with a higher frequency. In general, at lower spatial resolution, effort is evenly distributed over a large spatial unit. Within that unit, there will be high resolution spatial units with much higher frequencies, whereas other high resolution spatial units have lower frequencies or are not fished at all. Because of this and the fact that every haul in a heavily fished area causes less mortality than a similar haul in an area that was hardly fished or pristine, the net effect of increasing spatial resolution should be a decrease in population mortality.

The slope of the trend over time in population mortality decreases with increasing overlap: for an encounter mortality of 80% and low spatial resolution, this is −0.4% at low overlap to −1.8% at high overlap; at high spatial resolution it is +1.8% at low overlap to +0.1% at high overlap (Table 2). These large differences show that within an EAFM, target levels for mortality can only be set realistically at the highest level of information content, and that when mortality needs to be calculated (e.g. for non-target species), the spatial resolution is important. It is therefore proposed that a standard spatial resolution be used and, as long as this is not achieved, the spatial resolution should at least be reported whenever frequencies (Level 3) or calculated mortality values (Level 4) are used.

Determining the encounter mortality of a species in the gear used by a specific métier may be difficult, because almost every vessel will fish with gear with slightly different characteristics and rigging (e.g. for a beam trawl, the mesh size, the number of tickler chains, the use of a chain mat or flip-up rope), which could affect encounter mortality. It might seem appropriate therefore to distinguish each of these gear types, but the amount of work required to estimate encounter mortality would then be prohibitive. We suggest that a limited number of métiers be distinguished with standard gear types assumed for which encounter mortality can be determined.

The consequences of the above for the use of pressure indicators as part of an EAFM is that the best pressure indicator (i.e. annual fishing mortality) will differ between species, both in terms of its absolute value and its trend over time, and that this in turn may respond differently to management measures than the lower level pressure indicators. Considering this difference between species, the pressure expressed as annual fishing mortality of one or two commercial species will not be representative for the whole community. Therefore, if management objectives are set for community level indicators such as mean weight, mean maximum length, or biodiversity (Piet and Jennings, 2005), the community mortality will need to be determined as an integral of all the population mortalities that make up the community.

Here we have illustrated the process for a number of "virtual populations" of marine species, one that suffers 20% mortality in a beam trawl gear and the other 80%. Ultimately, it should be possible to calculate Level 4 indicators for real populations and communities using distribution maps of benthos and demersal fish (for example, see www.mafcons.org) and incorporating characteristic species or taxon-level information on the encounter mortality of the species making up the community in a given fished area. This is likely to be more achievable in the near future for benthic invertebrates, because distribution maps often exist for both fish and benthos, but encounter mortality has only been determined for some of the common benthic species (for review and meta-analysis, see Collie, 2000, and Kaiser *et al.*, 2005), using sampling techniques that allow the determination of absolute pre- and post-haul abundances (see example for a limited number of benthic species in Piet *et al.*, 2000). For fish, there are as yet no estimates of the encounter mortality caused by the passing of the gear despite the fact that in fishery science, catchability and gear efficiency are known concepts (e.g. Dickson, 1993). To further complicate the matter, encounter mortality consists not only of mortality of animals caught in the net (i.e. catchability), but also mortality through contact with the gear (e.g. after passage through the net). This is more relevant for benthos than it is for fish (Bergman and van Santbrink, 2000).

General discussion

The usefulness in an EAFM of this framework of pressure indicators becomes apparent when the minimum level of information content required to evaluate a particular type of management measures can be identified. Effort control through decommissioning is already reflected at the lowest level, while it will show at the second level if it is implemented through a reduction in days at sea. Technical measures may show up at the third level if it involves changes in the gear characteristics that determine the area fished in relation to effort (e.g. if the width of a beam trawl is reduced), but usually only affect the encounter mortality, and will therefore only show up at Level 4. Spatial measures such as marine protected areas can only be evaluated at Level 4.

This case study demonstrates that it is possible to develop pressure indicators that describe the impact induced by fishing activities on a particular ecosystem component or system (e.g. the demersal system here), and which are appropriate for use as part of an EAFM. However, the best pressure indicators also come with extensive data requirements that at present are only marginally available even in one of the most data-rich marine environments in the world. An EAFM can only be successfully implemented and monitored if the fishing pressure can be described at a level of information content that is adequate to guide management decision-making.

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The physiological status and mortality associated with otter-trawl capture, transport, and captivity of an exploited elasmobranch, *Squalus acanthias*

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Mandelman, J. M., and Farrington, M. A. 2007. The physiological status and mortality associated with otter-trawl capture, transport, and captivity of an exploited elasmobranch, *Squalus acanthias*. ICES Journal of Marine Science, 64: 122–130.

To assess the physiological responses and associated mortality in spiny dogfish (*Squalus acanthias*) following capture in an otter trawl and exposure to additional conditions, blood samples were obtained subsequent to three sampling intervals: capture (T1), transport (T2), and captivity (T3). The results indicate that marked differences existed in blood chemistry at each sampling interval. Acid–base parameters (vascular pH, pO_2, pCO_2), serum Ca^{2+} and Cl^-, and haematocrit were maximally disrupted at T1, but progressively resolved to presumed basal values by T3. Concentrations of whole-blood lactate, plasma total protein, additional sera electrolytes (Na^+, K^+, Mg^{2+}), and BUN (urea) were maximally compromised at T2, but also recovered by T3. In contrast, serum glucose levels were similar at T1 and T2 but rose to peak levels by T3. Although blood parameters were substantially altered, dogfish mortality was low (2 out of 34; 5.9%), suggesting a strong degree of resilience to compounded stressors associated with capture, transport, and captivity.

Keywords: blood chemistry, captivity, mortality, spiny dogfish, stress, transport, trawl.

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Introduction

The spiny dogfish (*Squalus acanthias*) is a coastal squaloid with a range extending from Labrador to Florida in the western Atlantic (Sosebee, 2000) and a circumboreal global distribution (Nammack *et al.*, 1985). Like most elasmobranchs, dogfish exhibit K-selected life history characteristics, which include slow growth, late maturity, and low fecundity (Nammack *et al.*, 1985; ASMFC, 2002). Female dogfish also display a prolonged (18–22 month) period of gestation and when a directed fishery exists for the species, are selected over males as a function of their larger maximum body sizes at maturity (Sosebee, 2000). Primarily because of these factors, heightened fishing pressure in the western North Atlantic during recent decades led to a reported 75% decline in mature female stocks between 1998 and 2003, and a concomitant scarcity in recruitment (ASMFC, 2002; NEFSC, 2003). Incidental capture of dogfish is also extensive. Low trip limits and limited commercial value have led to the discarding of consistently large quantities of dogfish in western North Atlantic commercial fisheries. Therefore, post-capture condition and discard survivability of dogfish hold major implications regarding stock health and associated management. Augmenting the assessment of capture stress with an investigation of transport and captivity can provide insight regarding a species' capability to recover following a particular form of capture and a variety of additional stressors, an important factor when assessing how resilient populations are when captured and discarded as bycatch.

Although many studies have investigated the physiological responses to capture, handling, transport and confinement stressors either individually or collectively in teleosts (e.g. Barton *et al.*, 2003; Sulikowski and Howell, 2003), fewer have done so in elasmobranchs (Cliff and Thurman, 1984; Torres *et al.*, 1986; Smith, 1992). Moreover, no investigation to date has addressed the physiological threshold of dogfish related to the rigours of catch and release, and to our knowledge, no study has documented the post-capture physiological implications of mobile-fishing capture in an elasmobranch. In order to gain greater understanding of physiological stress responses and the resilience of dogfish, a sample of trawl-captured dogfish was transported, held captive for 30 d, and assessed for physiological status and mortality following the completion of each study phase.

Material and methods

Animal collection, transport, and holding in tanks

Dogfish were caught in six, 45 min, moderately packed (~270–300 kg) otter trawls during two consecutive days (2 and 3 September 2004) southeast of Chatham Inlet (41°38′N 69°48′W) aboard the commercial fishing vessel "Joanne A III" (Chatham, MA, USA). A 350 hp, semi-high-rise Danish otter trawl containing 302 meshes in the fishing circle and a 15.2 cm mesh was used. The net also possessed 15.0 fathom top and bottom legs and 20.0 fathoms of ground cable. The trawl doors weighed 454 kg. A hard-bottom sweep on the bosom section was utilized to avoid boulders. The depth of trawling ranged between 50 and 65 m on a cobble and sand seabed, with 13.0–14.0°C bottom-water temperature. The total length (TL) of dogfish utilized in the study