Spatio-temporal patterns of fishing pressure on UK marine landscapes, and their implications for spatial planning and management

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The spatio-temporal distribution of fishing pressure on marine landscapes in offshore UK (England and Wales) waters is assessed, based on a time-series of fishing vessel monitoring system (VMS) data for UK and foreign fleets deploying beam and otter trawls, and scallop dredges. The results reveal that marine landscapes with coarse or mixed sediments and weak or moderate tide stress are heavily fished. Marine landscapes experienced different intensities of fishing pressure depending on their spatial location in UK offshore waters and the regional heterogeneity of landscape types. Spatial patterns of fishing pressure vary by region, but within regions, patches of high fishing pressure remain centred at the same locations. When designing marine management plans, it is important to take account of the spatial extent and patchiness of fishing activity, and the consistency with which areas are fished in the same region from year to year. Descriptions of the spatial distribution of fishing pressures will become more meaningful at a local level if they also reflect the sensitivity of the habitats to those pressures. The further development of such sensitivity analyses, using life-history traits or measures of benthic production, is now becoming a priority.

Keywords: GIS, marine spatial planning, semi-variogram, spatial autocorrelation, VMS.

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Introduction

The expansion of offshore activities, such as hydrocarbon exploration and production, the generation of wind and wave energy, and fishing, has led to increased development pressures on the marine environment (Defra, 2005). The increase of human pressures results in an increase in complexity of spatial use, requiring the protection of threatened and declining habitats and species (Douvere and Ehler, 2007). In the past, marine management approaches have been sectoral rather than resolving multiple-use conflicts. Therefore, in recent years, emphasis has been placed on an ecosystem approach to natural resource and environmental management (Douvere and Ehler, 2007). In Europe, the Maritime Strategy aims to develop integrated marine management which maintains ecosystem health while ensuring appropriate use of the marine environment for current and future generations (Rice *et al.*, 2005). This requires integrated planning tools such as strategic assessment, coastal-zone management, and marine spatial planning for regulating, managing, and protecting the marine environment (Tyldesley, 2006; Boyes *et al.*, 2007; Douvere *et al.*, 2007). Crucial to successful marine spatial planning is the accurate assessment of spatial distribution of human activities and their associated pressures (Defra, 2005). For UK offshore waters, Eastwood *et al.* (2007) quantified the footprint of various human activities during a single year (2004). However, to develop comprehensive marine spatial planning, the footprint of human activities, their intensity, and associated spatio-temporal variability need to be quantified to assess uncertainty and to draw sound conclusions regarding their impact on the marine environment.

Another important consideration for planning is the need to describe the spatial extent of seabed features and habitats (Boyes *et al.*, 2007). Mapping of marine landscapes is important in determining the nature of biological communities and may assist in the designation of potential marine protected areas (MPAs) or areas of high biological value within a marine spatial planning framework (Roberts *et al.*, 2003; Connor *et al.*, 2006; Boyes *et al.*, 2007). Connor *et al.* (2006) classified the marine landscapes of the UK seabed based on topographic and physico-geographic characteristics. In the Irish Sea, Boyes *et al.* (2007) used these marine landscapes to test a proposed marine spatial plan. Significant to a sound marine spatial framework is the quantification of human pressures by marine landscapes, but as yet this link has not been explored empirically. The most important human pressure in terms of its spatial extent and level of impact on the UK marine environment results from fishing (Collie *et al.*, 1997; Rijnsdorp *et al.*, 1998; Dinmore *et al.*, 2003; Eastwood *et al.*, 2007). As a consequence, the development of a marine spatial planning framework in UK waters requires detailed understanding of the spatial and temporal variation of fishing pressure at the spatial scale of both regions and marine landscapes.

In nature, living beings are distributed neither uniformly nor at random, but are aggregated in patches or other types of spatial structures (Legendre and Fortin, 1989). Aggregations in the observed distribution of fish can be caused by size and age class (Wieland and Rivoirard, 2001), sampling period (Rueda and Defeo, 2001), or habitat association (Stelzenmüller *et al.*, 2005, 2007). The presence of a spatial structure in their distribution is indicated by spatial autocorrelation between pairs of samples that can be characterized and modelled mathematically by geostatistics (Cressie, 1991). As fishing activities are linked to the occurrence of targeted species (Swain and Wade, 2003), it is likely that data on fishing pressure are spatially structured, so data are spatially autocorrelated (Rijnsdorp *et al.*, 1998; Jennings and Cotter, 1999; Poos and Rijnsdorp, 2007). For the practical aspects of marine spatial planning such as the implementation of monitoring programmes, the characterization of spatial structures in fishing pressure is crucial to improving the design and interpretation of surveys and experimental studies, by relating sampling programmes to adequate scales of variation.

This study provides a detailed assessment of fishing pressure in offshore UK waters at the scale of both regions and marine landscapes by quantifying its uncertainty, which is well beyond the general mapping of human pressure footprints of a single year (see Eastwood *et al.*, 2007). We only considered fishing practices having a direct physical impact on the offshore seabed, such as beam and otter trawling, and scallop dredging (Eastwood *et al.*, 2007). We extracted all positional data of UK and foreign fishing vessels $\geq$18 m long only deploying beam or otter trawls or shellfish dredges from satellite-based vessel monitoring system (VMS) databases (Deng *et al.*, 2005; Murawski *et al.*, 2005; Davies *et al.*, 2007) for the years 2001–2006. Within a geographical information system (GIS) framework, we filtered and converted these positional data into measures of fishing pressure based on the VMS data time-series. With these data, we aimed to (i) assess the fishing pressure on UK marine landscapes at the scale of both regions and total area, (ii) characterize the spatial structuring of regional fishing pressure, and (iii) describe the implications of the results for spatial planning and management.

Material and methods

Study area and fishing activity data

The study area covered the waters of the UK (England and Wales, E&W), divided into reporting areas (RAs) 1–5 (Defra, 2005), describing the northern North Sea (1), southern North Sea (2), eastern English Channel (3), western English Channel, Celtic Sea, and southwest approaches (4), and the Irish Sea and North Channel (5; Figure 1). From the EC VMS database, we extracted for our study area positional data of relevant UK and foreign vessels (Belgian, Danish, Dutch, French, Irish, and Spanish) deploying beam or otter trawls and shellfish dredges and with a length of $\geq$18 m from 1 January 2001 to 31 December 2006. During this period, there were several changes in EC regulations (EC, 2003). From 2001, only vessels $\geq$24 m long transmitted

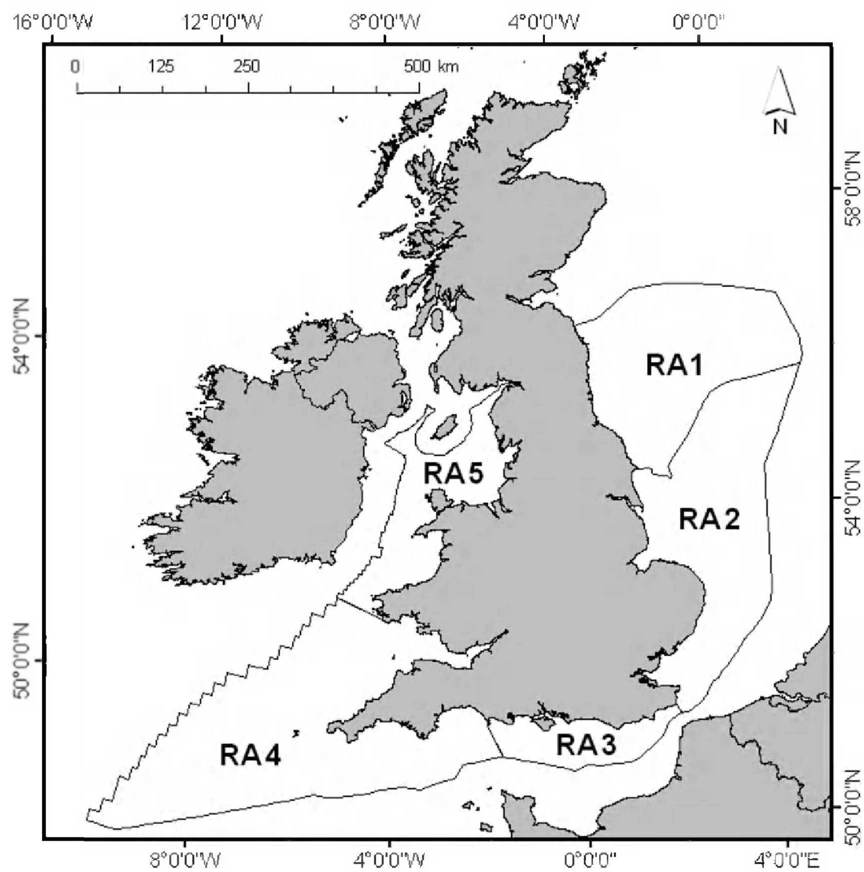

Figure 1. Study area covering the UK (England and Wales) waters, showing Defra reporting regions RA1, RA2, RA3, RA4, and RA5.

their positions, and in 2004 vessels $\geq$18 m and in 2005 vessels $\geq$15 m had to be included in the VMS database. Although we could select UK vessels by their length and gear type *a priori*, we needed to associate fishing gear information with foreign vessel observations by obtaining the respective information from the EC vessel registration database (http://ec.europa.eu/fisheries/fleet/index.cfm). Approximately 10–15% of the vessels in foreign fleets could not be associated with a gear type, so these positions were removed from subsequent analyses. As information on the length of foreign vessels was not available for much of the fleet, it was not possible to remove foreign vessels of length 15–18 m in 2005 and 2006. However, from the location of UK vessels <18 m, we assumed that very few foreign vessels of this size class were represented in the UK database, because such vessels often fish closer to the coast of the host nation. Further, in cases where information on vessel speed was not available (non-UK fleets before 2006; transmission errors), we calculated speed based on the distance and time between consecutive positional data.

Following the approach described by Eastwood *et al.* (2007), we applied speed rules to determine whether the vessels were fishing or steaming. Then, we used the straight-line approach to convert the remaining fishing locations into trawl tracks. In contrast to the approach of Eastwood *et al.* (2007) or Mills *et al.* (2007), we did not include estimates of uncertainty into the data by allowing for possible deviations of the vessels tracks from a straight line. To convert the fishing tracks into footprints, we buffered track lines of beam trawlers with 24 m (2×12 m) beams, otter trawler with 4 m (2×2 m) scour tracks left by trawl doors, and shellfish dredges with 20.4 m (24×0.85 m) dredge marks (Dinmore *et al.*, 2003; Eastwood *et al.*, 2007; Smith *et al.*, 2007). Further, we did not dissolve track boundaries to calculate the exact fishing footprints for each year, but retained all records of overlapping tracks. For the subsequent analyses, we used a grid with a 2×2 nautical mile (henceforth, mile) resolution to produce measures of total fishing pressure. This grid is a good compromise between the large spatial scale of the study area and the fine resolution of fishing positions shown by VMS (Eastwood *et al.*, 2007; Mills *et al.*, 2007).

Marine landscapes

A classification of the UK seabed into 44 types of marine landscape was undertaken by the UK Joint Nature Conservation Committee (Connor *et al.*, 2006) primarily using topographic and physiographic characteristics. We modified the marine landscape map of Connor *et al.* (2006), by converting the original UK marine landscape map to a coarser grid with a resolution of $\sim 2 \times 2$ miles. In cases where more than one marine landscape was associated with one grid cell, the dominant marine landscape category was selected (see Figure 2). For our analyses, we only used landscapes that contributed at least 1% to either the total UK (E&W) waters or to each region. The corresponding codes of the remaining 30 marine landscape categories are listed in Table 1.

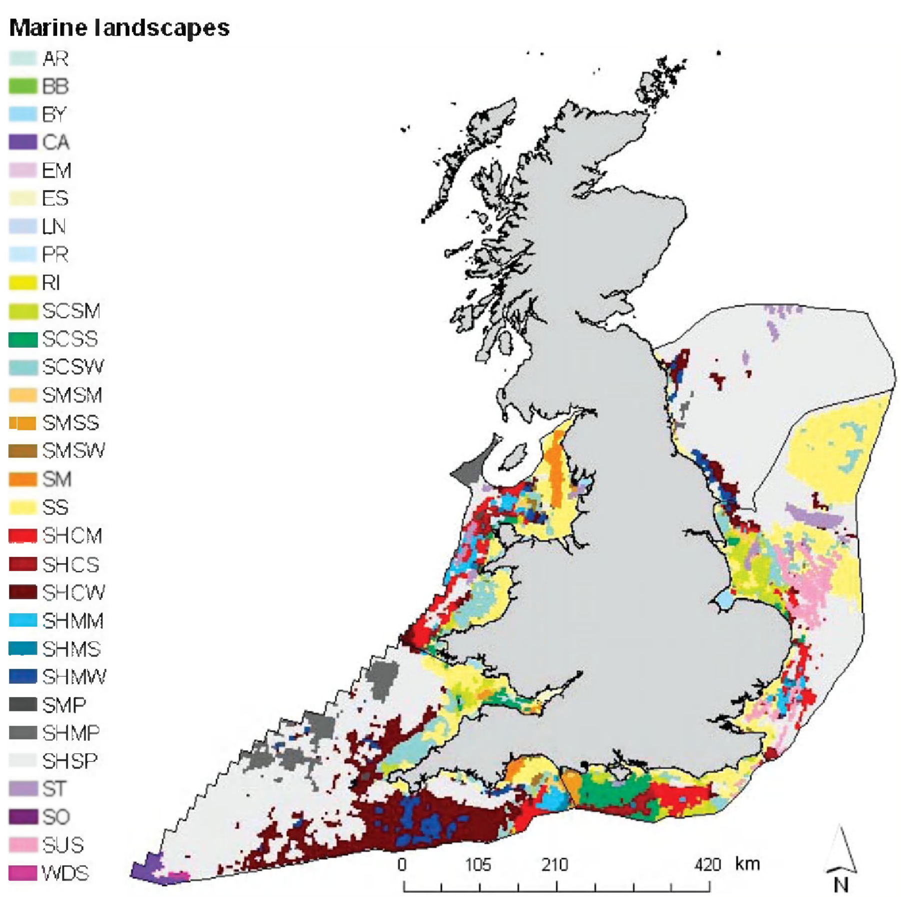

Figure 2. Marine landscape categories in UK (England and Wales) waters based on a 2×2 nautical mile grid resolution; modified after Connor *et al.* (2006).

Table 1. Codes for all marine landscape categories in UK (England and Wales) waters summarized by a grid resolution of 2 × 2 nautical miles.

Marine landscape category	Code
Aphotic rock	AR
Barrier beach	BB
Bay	BY
Canyon	CA
Embayment	EB
Estuary	ES
Lagoon	LN
Photic rock	PR
Ria	RI
Shallow coarse sediment plain – moderate tide stress	SCSM
Shallow coarse sediment plain – strong tide stress	SCSS
Shallow coarse sediment plain – weak tide stress	SCSW
Shallow mixed sediment plain – moderate tide stress	SMSM
Shallow mixed sediment plain – strong tide stress	SMSS
Shallow mixed sediment plain – weak tide stress	SMSW
Shallow mud plain	SM
Shallow sand plain	SS
Shelf coarse sediment plain – moderate tide stress	SHCM
Shelf coarse sediment plain – strong tide stress	SHCS
Shelf coarse sediment plain – weak tide stress	SHCW
Shelf mixed sediment plain – moderate tide stress	SHMM
Shelf mixed sediment plain – strong tide stress	SHMS
Shelf mixed sediment plain – weak tide stress	SHMW
Shelf mound or pinnacle	SMP
Shelf mud plain	SHMP
Shelf sand plain	SHSP
Shelf trough	ST
Sound	SO
Subtidal sediment bank	SUS
Warm deep-water sand plain	WDS

Temporal resolution of fishing pressure

To define an appropriate temporal resolution for the fishing tracks derived, we compared the within-year variability with the between-year variability in UK (E&W) waters. The dataset best suited to this purpose was a subset of the processed VMS time-series data, with complete information on gear type and vessel length (UK vessels ≥24 m deploying beam trawls, otter trawls, and scallop dredges; 2001–2006). We explored the effects of the factors quarter (4 factor levels), year (6 factor levels), and marine landscape category (21 factor levels) on these square-root-transformed total fishing pressure data, per grid cell, using one-way and two-way ANOVA.

Analysis of spatio-temporal patterns in fishing pressure

We explored the overall spatial and temporal variability of fishing pressure within the study area with the help of NMDS plots. For each year and region, we calculated the proportion of the marine landscape surface fished (%), using the complete VMS data time-series. We computed NMDS plots with Bray–Curtis dissimilarity matrices (for a detailed description of the methodology, see Clarke and Warwick, 1998), and standardized fishing pressure data with a Wisconsin double standardization, where fishing pressures are first standardized by their maxima then by site totals (Legendre and Gallagher, 2001).

We estimated the average annual fishing pressure ($AvAFP_{SU}$) and its standard deviation ($SDAFP_{SU}$) for each spatial unit (grid cell, marine landscape, region, and total area) for 2001–2006 as follows:

$$\mathrm{AvAFP_{SU}} = \sum \frac{\sum \mathrm{TFP_{year1}}}{\mathrm{SUA}} + \frac{\sum \mathrm{TFP_{year2}}}{\mathrm{SUA}} + \cdots + \frac{\sum \mathrm{TFP_{yearx}}}{\mathrm{SUA}} \text{ per number of years}, \qquad (1)$$

where TFP (m^2) is the total annual fishing pressure per cell, standardized by the spatial unit surface area (SUA; m^2). Hence, $AvAFP_{SU}$ represents the annual average proportion of a spatial unit affected by beam trawling, otter trawling, and scallop dredging. Based on this measure of fishing pressure, we mapped the average fishing pressure and its variation within the study area and by marine landscape for each Defra reporting region.

Spatial structuring of fishing pressure

We assessed the spatial structure of $Z(x)$ ($AvAFP_{cell}$) for each region using geostatistics. Empirical semi-variograms were used to describe the extent of spatial correlation between data $\gamma(h)$, measuring half the variability between data points (grid cell mid-points) as a function of their distance apart. In the absence of spatial autocorrelation between data, the semi-variance is equal to the variance of $AvAFP_{cell}$ of all observations. Where a significant trend was encountered, we detrended data (Kaluzny *et al.*, 1998), because long-range trends can mask or bias spatial structuring. Subsequently, we computed omnidirectional semi-variograms using the robust "modulus" estimator, which is supposed to be resistant to extreme values and skewed data distribution (Cressie, 1991):

$$\hat{\gamma}(h) = \frac{\left\{1/N(h) \sum_{x_i-(x_i+1)-h} \left|Z(x_i+h) - Z(x_i)\right|^{1/2}\right\}^4}{(0.914 + (0.988/N(h)))}, \qquad (2)$$

where $Z(x_i)$ is the realization of $AvAFP_{cell}$ at grid cell x_i, $Z(x_i+h)$ another realization separated from x by a discrete distance h (measured in m), and $N(h)$ the number of pairs of observations separated by h. In a second step, we fitted theoretical covariance functions to the empirical semi-variograms with the help of a weighted least-squares procedure (Cressie, 1991). We automatically fitted the parameters nugget (C_0), sill (C), and range (a) of spherical models (Cressie, 1991). Low values of the nugget parameter reflect a high degree of spatial autocorrelation in the data. The range parameter can be interpreted as the average patch diameter of the spatial phenomenon studied (Sokal and Oden, 1978). In other words, the range parameter reflects the distance from where data are no longer spatially autocorrelated. In addition, to measure the reproduction of data by the fitted models, we conducted cross-validation. Results of this jack-knifing method are given by standardized errors: with a mean (Z-score) of 0 and a standard deviation (s.d.–Z-score) of 1, models represent the data adequately (Isaaks and Srivastava, 1989).

Results

Temporal resolution of fishing pressure

We found significant between-year variation of fishing pressure measures [$F(5, 498) = 30.3$, $p << 0.05$, MSE = 0.07, $r^2 = 22.6\%$], and significant variation between marine landscape categories [$F(20, 438) = 27.91$, $p << 0.05$, MSE = 0.05, $r^2 = 51.7\%$]. In contrast, ANOVA results did not reveal significant fishing pressure variation between quarters [$F(3, 500) = 1.78$, $p = 0.15$, MSE = 0.0065, $r^2 = 0.5\%$]. As the factor quarter was not significant, we used for subsequent analysis the temporal resolution of years to summarize total fishing pressure. The interaction between the factors marine landscape and year was significant [$F(125, 378) = 22.53$, $p << 0.05$, MSE = 0.011, $r^2 = 84.25\%$]. Therefore, for this VMS time-series subset, the spatial component explained a greater proportion of data variation than the temporal component.

Spatio-temporal variability of fishing pressure patterns

NMDS plots of annual fishing pressure in each region, and total fishing pressure for each marine landscape (Figure 3), confirmed for the full dataset that spatial variability dominated over temporal variability as all regions built clear clusters, whereas all temporal replicates were located within these clusters. Dissimilarities between regions were caused by their respective composition of marine landscapes. Whereas the east coast (RA2) was characterized by the categories Bay (BY), Subtidal sediment bank (SUS), and Shelf trough (ST; Figure 3), the southwest (RA4) was characterized by Warm deep water sand plain (WDS). In contrast, Shelf mixed sediment plain–moderate tide stress (SHMM) occurred in each region and experienced similar levels of fishing pressure with time.

Values of the average annual fishing pressure ($AvAFP_{cell}$) were not homogenously distributed over the total study area (Figure 4a). The general increase of fishing pressure from coastal areas to offshore areas reflects the exclusion of large vessels from the coastal areas. The coefficient of variation ($CV = SDAFP_{cell}/AvAFP_{cell} \times 100$, %) of the average annual fishing pressure (Figure 4b) showed locations experiencing high and low temporal consistency of fishing pressure. A low CV (<50%) reflects a high level of consistency of high or low fishing pressure within a cell over time, a CV of 100% characterizes areas with high annual fluctuation around the mean fishing pressure, and a CV >150% defines areas where the fishing pressure was most inconsistent over time. Areas in the northeast of the UK showed the highest (CV <50%) and the lowest (CV >150%) levels of temporal consistency in fishing pressure. Approximately 50% (122 412 km^2) of the total area received a very low level of fishing pressure, whereas 2.5% (5963 km^2) of the total area had the highest level of fishing pressure (Figure 4c). More than 50% (135 469 km^2) of the total area experienced levels of high to medium consistency of fishing pressure over time (Figure 4d).

The variability of fishing pressure patterns in the total area (Figure 5, top left) and by region showed that the highest fishing pressure was on the marine landscape category Shelf coarse sediment plain–weak tide stress (SHCW), on average almost 30% of the surface of the marine landscape being affected by fishing annually, despite it being only the third largest landscape.

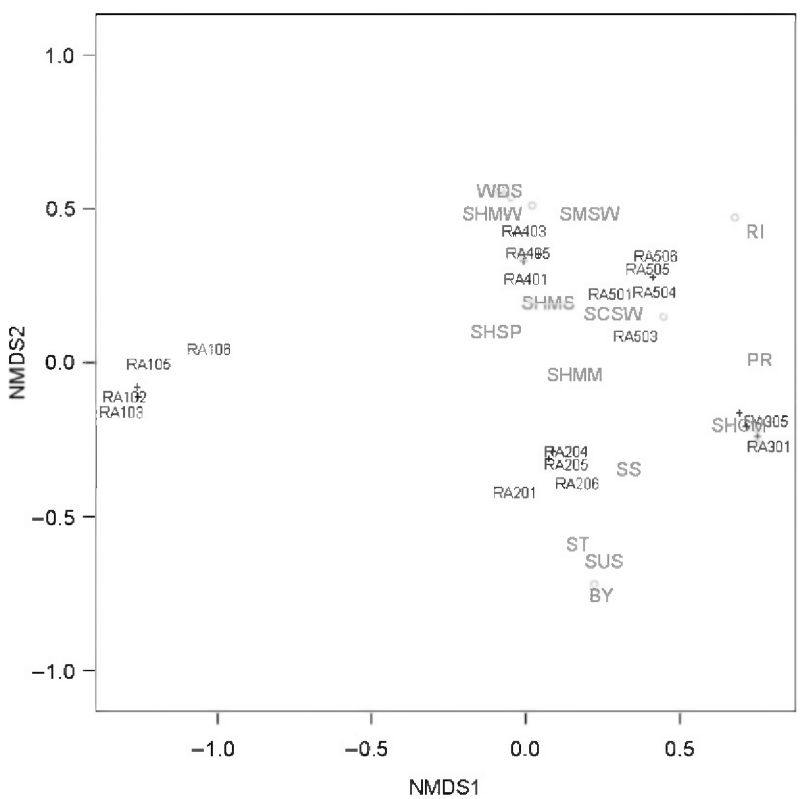

Figure 3. NMDS plot based on Bray–Curtis dissimilarities of annual fishing pressure on marine landscapes by region, reflecting the overall spatio-temporal variability of fishing pressure within UK (England and Wales) waters. Note: to avoid overlapping labels, temporal replicates have been replaced with a "plus" and marine landscape with a "circle".

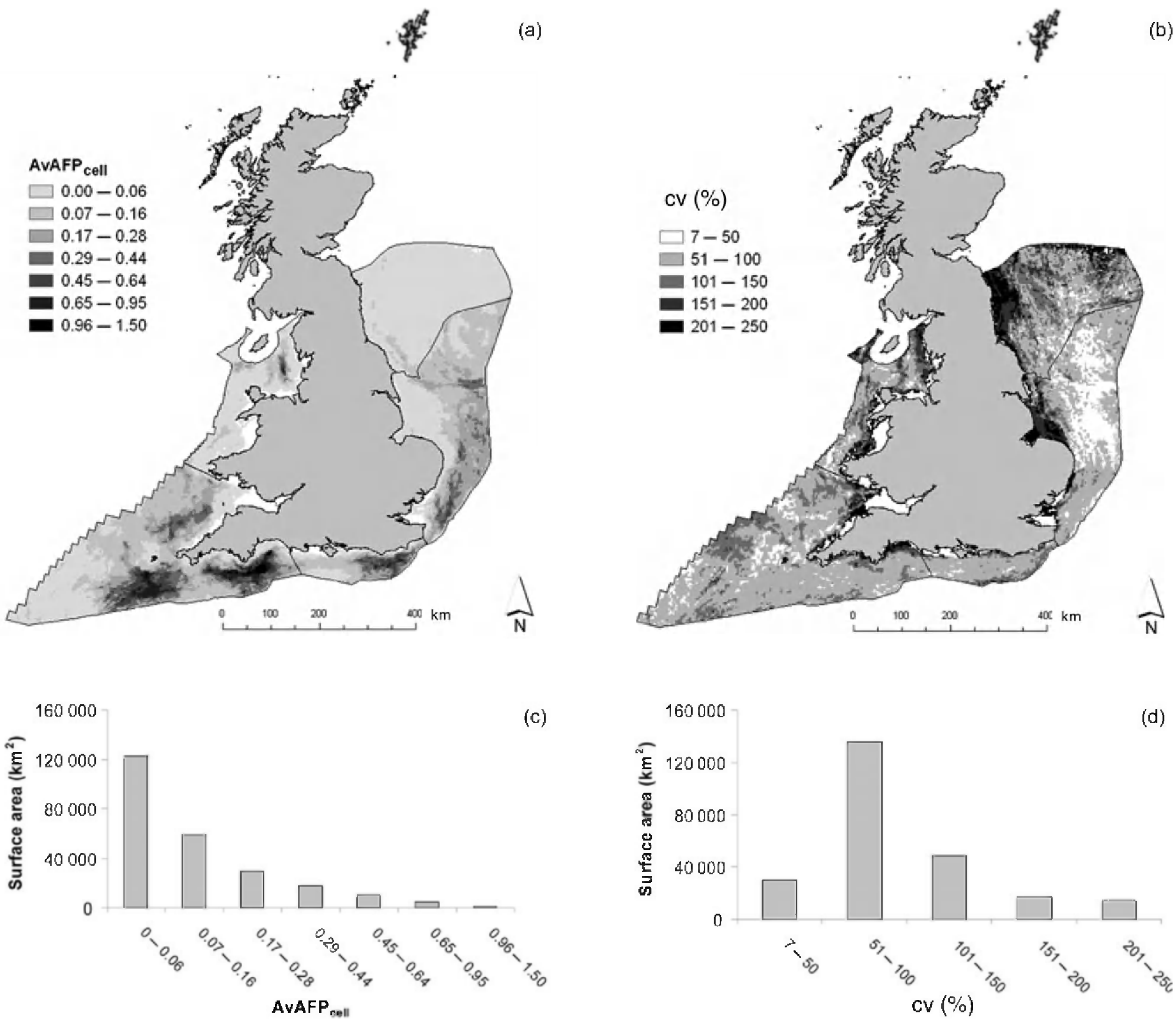

Figure 4. (a) Average annual fishing pressure ($AvAFP_{cell}$), as a proportion of grid cell (2 × 2 miles) affected by beam trawling, otter trawling, and scallop dredging; (b) its coefficient of variation [$CV(\%) = SDAFP_{cell}/AvAFP_{cell} \times 100$]. (c) Bar charts indicating the total surface (km^2) experiencing respective levels of fishing pressure, and (d) the temporal consistency in fishing pressure.

For each region, the rank of marine landscapes most affected by fishing varied. However, marine landscapes located at the shelf with weak or moderate tide stress were the categories most impacted by mobile fishing gears. We found in general a positive exponential relationship between the mean pressure value in each 2 × 2 mile cell ($AvAFP_{cell}$) and its variance ($VarAFP_{cell}$), suggesting that areas experiencing high pressure also had a high degree of variation over time (Figure 5, top left).

Spatial structuring of fishing pressure

We deployed experimental semi-variograms for the regional structural analysis of fishing pressure ($AvAFP_{cell}$; see Figure 6). These showed reduced semi-variances when the data were detrended with first order (RA5) and second order (RA1–4) polynomial trends. In all cases, data were spatially autocorrelated. Exploration of the data for different levels of patchiness of fishing pressure between the regions, using fitted range parameters 30–125 km (Table 2), identified small patches and therefore a higher level of patchiness in RA1 (38 km) and RA3 (30 km), whereas RA2 and RA4 were characterized by low levels of patchiness, with a dominating patch having an approximate diameter of 100–125 km (Table 2; see also Figure 4).

Discussion

Spatio-temporal patterns of fishing pressure

A temporal resolution of years reflected the general pattern of fishing pressure well, because the within-year seasonality of each of the component fisheries is compensated by aggregating them into one measure of annual fishing pressure. However, these values of mean fishing pressures and variances have to be interpreted with caution because they contain *a priori* variability from the inconsistency of VMS data quality because of changing EC regulations. In 2004, vessels of 18–24 m were included, in 2005 vessels of 15–18 m, and in 2006 EC regulations on satellite-tracking devices required improved speed transmission (EC, 2003), which led to a general increase in fishing activity in 2006

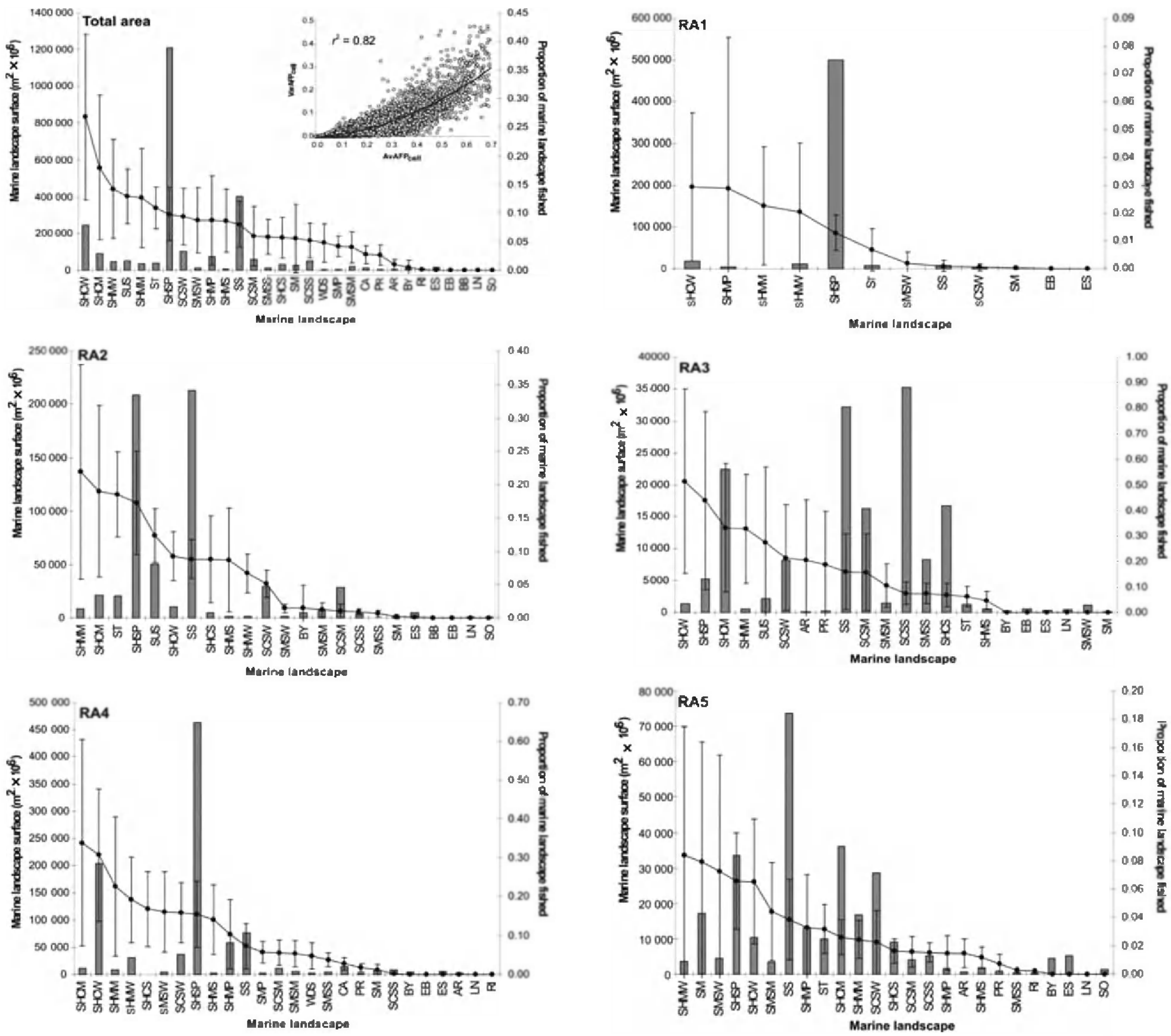

Figure 5. Average annual fishing pressure per marine landscape (AvAFP$_{\text{marine landscape}}$; shown as a continuous line along with its standard deviation, SDAFP$_{\text{marine landscape}}$), and the respective surface area ($m^2 \times 10^6$) of each marine landscape as bars, for the UK shelf and the five Defra reporting regions. In the right corner of the total area plot, the correlation between AvAFP$_{\text{cell}}$ and its variance (VarAFP$_{\text{cell}}$) is modelled with an exponential model, with r^2 denoting the percentage of variation in the dependent variable accounted for by the independent predictor variables.

as more positional data could be associated with a fishing activity. Although our time-series of VMS data is therefore biased towards increasing numbers of vessels reporting positional data and improved filtering of fishing activity, which probably led to an overestimation of the temporal variability of fishing pressure, we assume that for each year, the data reflected the actual spatial pattern of fishing pressure. However, across the spatial scales considered, we expect estimated absolute values of mean fishing pressure to be underestimated, because we used a straight-line approach to convert positional data into fishing footprints, which could underestimate the area impacted by fishing, vessels potentially deviating from this straight-line path when fishing (Eastwood *et al.*, 2007; Mills *et al.*, 2007). Also, we did not consider coastal fisheries performed by vessels <15 m.

The purpose of our study was to describe spatio-temporal patterns of fishing pressure rather than to produce exact quantifications of fishing pressure. We therefore consider the index AvAFP$_{SU}$ as a good qualitative indicator for fishing pressure of the fishing gears considered, which reflects true fishing pressure dynamics more reliably than estimates based on a single year of VMS data. To underpin this recommendation, we compared the temporal development of the VMS-based fishing pressure (FP) estimates within the total study area with the corresponding recorded gross tonnages (GT) of the same demersal fleets from the UK Sea Fisheries Statistics (www.statistics.defra.gov.uk; Figure 7). We adapted the GT records according to the changes in vessel length classes in the VMS database. In 2004, the increase of GT by the addition of the group of vessels having a length of 18–24 m was not matched by a corresponding increase in vessels transmitting their position. In contrast, VMS records of fishing pressure increased from 2004 to 2006, whereas overall tonnage declined (Figure 7). The deviation of time-series patterns

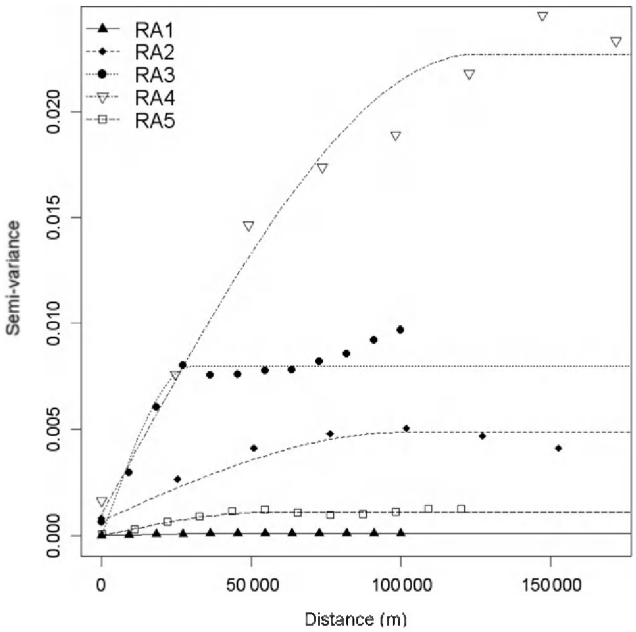

Figure 6. Semi-variograms and fitted spherical models of average annual fishing pressure ($AvAFP_{cell}$) for the regions RA1, RA2, RA3, RA4, and RA5.

Table 2. Estimated parameters nugget (C_0), sill (C), and range (a) of spherical models fitted to experimental semi-variograms of fishing pressure values ($AvAFP_{cell}$) for the reporting regions RA1–5, and the cross-validation results with the standardized mean error (Z-score) and its standard deviation (s.d.–Z-score).

Reporting region	Model	C_0	C	a (km)	Z-score	s.d.–Z-score
RA1	Spherical	0	0.0001	38	0	1.53
RA2	Spherical	0.0007	0.0042	100	0	0.84
RA3	Spherical	0	0.0080	30	0	0.96
RA4	Spherical	0.0010	0.0217	125	0	0.86
RA5	Spherical	0	0.0011	55	0	1.61

between the estimated fishing pressure and the respective fishing effort reflects the rate of change in VMS data quality attributable to changes in EU regulations (Defra, pers. comm.). For these reasons, only an average measure of fishing pressure and its variability should be used. As a result of recent technical improvements to the VMS system, it is likely that future analysis of VMS time-series data (from 2006 on) including various vessel size classes should be less biased.

Our exploratory analysis using NMDS showed that, for the total study area, the spatial variation of fishing pressure dominates over the temporal variation. Moreover, individual marine landscapes experienced different intensities of fishing pressure depending on their location. We observed that some marine landscapes representing soft seabed with weak or moderate tide stress such as Shelf coarse sediment plain–weak tide stress, Shelf sand plain, or Shelf mixed sediment plain–moderate tide stress supported up to 95% of the total fishing pressure, although their relative rank changed by region. These findings reflect the fishing gears considered for this study, because the use of bottom trawls is

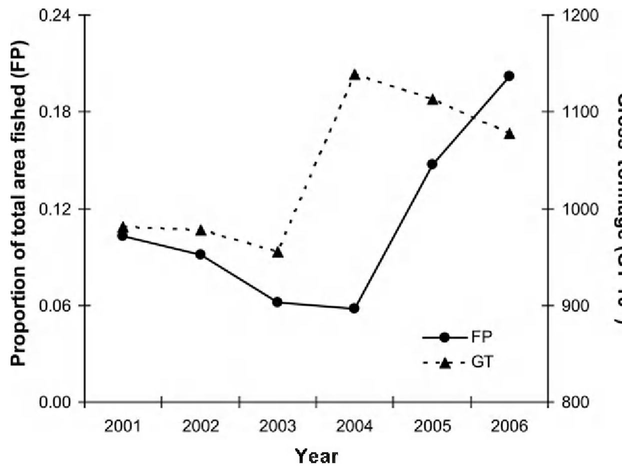

Figure 7. Comparison of the temporal pattern of estimated fishing pressure (FP) and the registered gross tonnages (GT) derived from the UK Sea Fisheries Statistics (www.statistics.defra.gov.uk). GT data include UK, Belgian, Danish, Dutch, French, Irish, and Spanish demersal trawlers $\geq$24-m long from 2001, $\geq$18 m from 2004, and $\geq$15 m from 2005.

restricted to areas with soft sediment where the chances of gear loss are small (Rijnsdorp *et al.*, 1998; Piet *et al.*, 2000). However, for this study, we standardized data resolutions by the use of grid cells, so we did not account for the small-scale complexity in marine landscape composition at scales smaller than 2×2 miles. As a consequence, our estimates of fishing pressures on marine landscapes are scale-related, something that has to be taken into account when interpreting these results.

In general, we found that areas with a medium level of temporal consistency in fishing pressure had relatively high fishing intensity. The coastal areas with the lowest level of temporal consistency in fishing pressure reflected rather occasional fishing activities by the demersal fleets within the past 6 years. The spatial pattern of temporal consistency of fishing pressure suggested that fishing pressure in highly fished areas tended to increase or decrease from year to year in patches at fixed locations, rather than to distribute evenly in space. This results in a consistent spatial distribution pattern of fishing pressure over time, and a considerable temporal variation of fishing intensity at the spatial scales of cells, marine landscapes, or regions. Our findings are in line with the results of Larcombe *et al.* (2001), who reported for Australian waters the greatest interannual variation in trawling patterns on grounds fished heavily. Further, Bellman *et al.* (2005) found significant year-to-year variation in bottom trawling on the fishing grounds off Oregon, USA. Besides inconsistency in data quality, possible reasons for interannual shifts in fishing pressure could be changes in target species, trip limit regulations, and individual fishing strategies (Babcock and Pikitch, 2000; Bellman *et al.*, 2005).

With respect to the spatial scale of our grid and respective regions, we observed for all regions high levels of spatial autocorrelation between cell-based fishing pressure measures. As nugget values were very low, almost all variability in the data could be attributed to a patchy distribution of fishing activity, in line with other studies showing highly heterogeneous distributions of fishing activities in the North Sea (Rijnsdorp *et al.*, 1998; Jennings and Cotter, 1999; Dinmore *et al.*, 2003), Baltic Sea

(Nilsson and Ziegler, 2007), Australian waters (Larcombe *et al.*, 2001), Pacific (Bellman *et al.*, 2005), and around Iceland (Ragnarsson and Steingrimsson, 2003). We expect the regional marine landscape composition to be mainly responsible for the patchy distribution of fishing pressure we observed, because the distribution of fish and shellfish resources will be closely governed by the structure of the demersal habitat.

Implications for marine spatial planning and management

The results of this study suggest that it is important when designing marine management plans to take account of the spatial scale of the major pressures acting on the environment. In this example, the patchiness of fishing activity, and the consistency with which fishing occurs in the same region year on year, suggests that this should be a strong guiding factor in the selection of the spatial extent of marine planning units.

Our results also show that a single marine landscape experiences different intensities of fishing pressure, depending on its spatial location in UK offshore waters. This is partly because the regional distribution of fishing pressure is related more to combinations of marine landscapes than to a single landscape category. The regional biogeography of the UK will also have an influence, because a marine landscape type off the southwest coast may not support the same fish and shellfish populations as the same landscape off the northeast coast. These results suggest that it will be necessary to take account of national distribution of human pressures by landscape when developing regional plans, and when identifying areas within which the fishing industry is the dominant marine sector.

These patchy and varied effort distributions will also influence the location and spacing of MPAs designed to protect rare, threatened, and representative UK seabed fauna (Defra, 2006). Although closed-area policy is still developing in northern Europe, it is likely that the location of human pressures and their consequences for seabed ecology and biodiversity will to some extent influence the MPA network design (Defra, 2006). This will act either by the protection of relatively unimpacted areas with low fishing pressure or by the designation of closed areas at heavily fished sites to protect fish and shellfish stocks (Dinmore *et al.*, 2003; Blyth-Skyrme *et al.*, 2006; Nilsson and Ziegler, 2007). In either case, the implication of the spatial patterns in fishing pressure distribution observed suggest that biological meaningful spacing of MPAs will vary between regions, because relevant spatial scales in fishing pressure vary by region. The relationship between areas of high fishing pressure and other neighbouring landscapes also needs to be considered when planning spatial management measures. Without concurrent reductions in total fishing capacity, the exclusion of potentially damaging fishing activities from one region will result in an unpredictable "domino-effect" as fishing activities would be displaced to the bordering landscapes, or the same landscape in other regions, with varying intensity. This suggests that the designation of such spatial management measures in a marine spatial plan should be undertaken at both a national and a regional scale. This is in line with the results of a marine spatial planning pilot project in the Irish Sea showing that marine spatial planning should be implemented on a regional scale to account for the respective ecosystem features (Vincent *et al.*, 2004; Boyes *et al.*, 2007).

The spatial autocorrelation in the data at this spatial resolution, associated with the patchy nature of fishing pressure, also suggests that this is an important consideration in management planning. From a practical perspective, it will be necessary to select a spatial scale (i.e. grid size of map) that takes into account the variability in all pressures of human activity, including that of fishing activity.

Although patches of high fishing intensity were centred at the same locations, our results show that their boundaries can vary from year to year through shifts in intensity. This suggests that temporal variability of fishing pressure derived from VMS data should be taken into account regardless of the spatial scale considered.

These descriptions of the spatial distribution of fishing pressures are useful for high-level management and planning, but to be meaningful at a local level they also require an understanding of the sensitivity of the habitats to different pressures (Gundlach and Hayes, 1978; Bremner *et al.*, 2005; Tyler-Walters *et al.*, in press). An assessment of the sensitivity of marine habitats to bottom trawls has been developed by MacDonald *et al.* (1996) and Hiscock (1999), who ranked benthic species by their likely sensitivity to the physical impacts of mobile fishing gear, based on species' physical characteristics and recoverability. The further development of such sensitivity analyses for the extensive offshore UK soft sediment plains will lead to sound impact assessments of all fishing activities on the marine environment (Hiddink *et al.*, 2007), because a high level of fishing pressure on a habitat does not necessarily lead to a great impact if the habitat is not sensitive. The use of species sensitivities, using life-history traits, and generic measures of size-based benthic production are two methods currently being developed to address this important issue.

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