Vol. 411: 259–270, 2010
doi: 10.3354/meps08662

MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser

Published July 29

# Recovery of UK seabed habitats from benthic fishing and aggregate extraction — towards a cumulative impact assessment

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ABSTRACT: Assessing cumulative impacts of multiple pressures on the marine environment can help inform management response. This requires understanding of the spatial and temporal distribution of human pressures and their impacts. Quantifying seabed recovery rates from 2 significant pressures in European waters, benthic fishing and aggregate extraction, is a significant step towards assessing sensitivity and cumulative impacts. Vessel monitoring system data were used to estimate the distribution and intensity of benthic fishing in UK (England and Wales) marine waters (2006 to 2007). Data were separated by towed bottom-fishing gears (scallop dredges, beam and otter trawls) and linked to habitat in a geographic information system. Recovery periods of seabed habitats were estimated by literature review, for gear types and fishing intensity. Recovery rates generally increased with sediment hardness, and habitats required longer periods of recovery from scallop dredging than from otter or beam trawling. Fishing pressure across the habitat–gear combinations was such that 80% of the bottom-fished area was estimated to be able to recover completely before repeat trawling, based on mean annual trawl frequencies. However, in 19% of the UK's bottom-fished seabed, scallop dredging in sand and gravel and otter trawling in muddy sand and reef habitats occurred at frequencies that prevented full habitat recovery. In 2007, benthic fishing and aggregate extraction occurred together in an estimated 40 km$^2$ (<0.02%) of the UK seabed. Cumulative impacts were estimated as total recovery time under 4 scenarios: greatest, additive, antagonistic and synergistic impacts. Recovery from aggregate extraction required much greater periods than from benthic fishing, and gravel was identified as a more sensitive habitat than sand.

KEY WORDS: Towed bottom-fishing · Recovery · Cumulative effects · Fishing intensity · Marine habitats · Aggregate extraction · UK seabed · Vessel monitoring system



## INTRODUCTION

Marine ecosystems provide a range of goods and services of value to humans (e.g. Costanza et al. 1997, Foley et al. 2010). Human activities can alter ecosystems, changing energy flows and biological communities, potentially leading to regime shifts (e.g. Choi et al. 2004, Hinz et al. 2009) which could undermine the supply of marine resources. Sustainable human use of marine resources requires an integrated approach to marine management, with knowledge of the location of activities and habitats, and accurate assessment of the condition of the marine environment. Mapping the distribution and intensity of pressures on different habitats and assessment of cumulative effects are relevant for spatial planning and conservation objectives, as required under legislation such as the Marine and Coastal Access Act (HMSO 2009) and the European Marine Strategy Framework Directive (EU 2008).

At present there are gaps in our knowledge and understanding of the impacts from major human activities and the interactions between them (OSPAR Commission 2003, Eastwood et al. 2007, Borja et al. 2008, Foden et al. 2008, Halpern et al. 2008b). Such inter-

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actions can create cumulative impacts, which need to be understood and quantified for more informed management decisions. Understanding cumulative impacts may be particularly important as repeated impacts may, for example, limit the ability of an ecosystem to recover over time. Furthermore, the combined impacts of many different activities might lead to different ecosystem responses from those of just one type. Cumulative effects can be comparative, additive or interactive in nature. A comparative effect predicts the overall impact to be equal to that of the single worst or dominant stressor (Bruland et al. 1991). Additive effects are simply the sum of the individual pressures (Halpern et al. 2008a), which is judged to be the case for pairs of pressures (Crain et al. 2008). Interactive effects are the results of multiple activities accumulating non-linearly, i.e. causing lesser (antagonistic) or greater (synergistic) effects than the sum of their parts (Smit & Spaling 1995, Folt et al. 1999, Cefas 2001, Oakwood Environmental 2002, Crain et al. 2008).

Recent ecological studies and meta-analyses have begun to address the theories and concepts of cumulative effects caused by multiple human activities on the seabed (e.g. Vinebrooke et al. 2004, Darling & Côté 2008, Halpern et al. 2008b). However, there have been fewer attempts to robustly quantify those impacts, and there is a gap in knowledge regarding environmental sensitivity and response to multiple pressures (e.g. Halpern et al. 2008a). Investigations of seabed recovery rates following disturbance provide a method of quantitatively estimating habitat sensitivity (Desprez 2000, Cooper et al. 2007, Foden et al. 2009). If a pressure occurs too frequently for a habitat to recover, the biomass and productivity of the benthic community decline (Hiddink et al. 2006a) and sustainability may be jeopardised. By geospatially modelling the process of cumulative impacts of 6 different pressures, using empirical data, Stelzenmüller et al. (2010) made an important advancement in the practical application of cumulative impact concepts. Using 2 major pressures currently affecting the UK seabed, we aimed to put these concepts into practice using quantitative data on habitat sensitivity and the effects of individual and cumulative impacts.

Whilst oil and gas extraction, wind farms, dumping, pipelines and cables can be important site-specific pressures acting on the UK (England and Wales) seabed, the major sources of seabed disturbance are near-bed currents, wind-induced waves (Hall 1994), aggregate dredging for mineral resources and bottom-trawling for fish (Jennings & Kaiser 1998, Eastwood et al. 2007, BMAPA 2008, Stelzenmüller et al. 2008). The UK is one of the largest producers of marine aggregate in the world, and 23.1 million tonnes of sand and gravel were extracted from the English and Welsh seabed in 2007 (BMAPA 2008). Recovery rates of the benthic community after cessation of disturbance by marine aggregate dredging have been used as a proxy of habitat sensitivity (Desprez 2000, Cooper et al. 2007, Foden et al. 2009). Equivalent examinations of impacts caused by other sectors will be required to fully assess habitat sensitivity and to estimate the cumulative effects of all pressures operating on the UK seabed.

A broad range of approaches to integrated assessments and cumulative impact assessments have been used, with varying degrees of success (Foden et al. 2008). We have adopted a sector by sector approach to analysing human pressures on UK seabed habitats, with a view to combining these in a multi-sectoral integrated assessment, and the work presented here contributes to this. The intention is that our approach will help inform decision makers, facilitating an ecosystem approach to marine management.

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). Taking an ecosystems approach in the context of fisheries management requires, inter alia, information on the response of marine habitats to fishing (FAO 2003). We considered fishing practices that have a direct physical impact on the seabed such as trawl and dredge gears (Rijnsdorp et al. 1998). Bottom-fishing directly affects the seabed and benthic communities and has been recorded as occurring in 75% of the global continental shelf (Kaiser et al. 2002). The degree of disturbance is dependent on 3 main factors: the type of fishing gear deployed, the intensity of fishing activity and the sensitivity of the habitat (Collie et al. 2000, Kaiser et al. 2006).

Benthic trawl gears can generate tracks on the seabed of 1.5 to 12 m width and 0.01 to 0.6 m depth, depending on sediment and gear type (Churchill 1989, Hall et al. 1990, Nédélec & Prado 1990, Hall 1994, Vanstaen et al. 2008). These gears are generally associated with detrimental impacts on the marine benthos (Currie & Parry 1999, Shephard et al. 2009). For example, to increase fish catches, beam trawls are usually fitted with chains designed to penetrate the top few centimeters of the substratum, and infauna and epifauna can be damaged as the chains pass through the sediment (Kaiser & Spencer 1994, 1995). Typically, shellfish dredges and flatfish beam trawls disturb the seabed more intensively than otter trawls (Hall 1994).

Biomass, abundance and cover of macro-fauna and -flora in sand and gravel environments are known to be negatively correlated with trawling intensity (Svane et al. 2009). In previous work, responses and recovery rates of different habitats to towed bottom-fishing gears have been reviewed (Kaiser et al. 2006, Pitcher et al. 2009). Kaiser et al. (2006) found that beam trawling and scallop dredging had significant negative, but short-term, impacts on biota in the highly energetic, shallow, soft-sediment habitats of sand and muddy sand sediments, with abundance recovering in <50 d, whereas in more stable gravel sediments, taxa were still ~40 % reduced after 50 d. Collie et al. (1997) found recovery from scallop dredging in shallow gravel sediment at Georges Bank to take 5 to 10 yr, and in the most stable environment of biogenic reefs, biota showed no signs of recovery after >4 yr.

Vessel monitoring system (VMS) data have been used successfully for estimating the spatial and temporal distribution of fishing effort (Deng et al. 2005, Murawski et al. 2005). This study analysed 2006 and 2007 VMS data for UK and foreign vessels deploying bottom-fishing gear in the UK (England and Wales) reporting region. The 4 objectives were to: (1) quantify the spatial extent and annual intensity of benthic fishing activity in 2006 and 2007 at a high resolution; (2) use data on recovery periods from published, peer-reviewed studies to estimate the proportion of fished habitats in which recovery would be possible at recent (2007) levels of fishing effort; (3) combine the results with published spatial data on aggregate dredging in UK waters (Foden et al. 2009) in order to investigate where the 2 pressures coincided during 2007, potentially giving rise to cumulative impacts on the seabed; and (4) estimate overall recovery times for the 4 cumulative effects scenarios proposed by Stelzenmüller et al. (2010) — greatest, additive, antagonistic and synergistic. The latter allowed us to determine the sensitivity of the findings to different measures of impact estimation, for different habitat types. Our key aims are for this information to contribute to an integrated assessment of cumulative impacts from simultaneous pressures acting on the UK seabed, and to help in the application of an ecosystem approach to marine management.

## MATERIALS AND METHODS

**Study area and VMS data.** The study area comprised the marine waters of the UK (England and Wales). Regions were delineated using boundaries determined for environmental status reporting under Charting Progress (Defra 2005).

Satellite-based VMS have been fitted to fishing vessels ≥15 m since 2005 to automatically record identity, position, speed and heading data at an average frequency of ~2 h. Vessels exclusively fishing within 12 nautical miles (n miles) of the shoreline or that undertake fishing trips of less than 24 h are exempted (Dann et al. 2002). Positional data of UK and European vessels (Belgium, Denmark, Holland, France, Ireland and Spain) in the study area for the 2 yr period covering 2006 and 2007 were made available by the Defra Marine Fisheries Agency. Speed rules were applied to determine fishing or steaming activity (Lee et al. in press). Benthic fishing gears were identified using UK logbook data or the European vessel register, and these were aggregated to 3 bottom-fishing gear types: beam trawls, otter trawls and dredges.

**Swept area of fishing gears.** The swept area of trawls or dredges depends on the width of that part of the gear which is in contact with the seabed. Three main bottom-fishing gear types are deployed in UK waters: beam trawls, scallop or hydraulic dredges and otter trawls. A subset of the latter includes *Nephrops* (prawn) trawls. Beam trawlers generally operate twin rigs with an overall width of 24 m (Rijnsdorp et al. 1998, Dinmore et al. 2003), and we used this width for calculating the swept area.

Scallop dredges are often deployed from beam trawl vessels in the UK, with a maximum number of 9 dredges (~1 m wide) per beam. As 2 beams are generally deployed, 18 m was used as an estimate of width for calculating the swept area. The use of hydraulic suction dredges for fishing shellfish is geographically restricted in UK waters. Mechanised dredges of this nature generate tracks on the seabed up to 3 m wide (Hall et al. 1990, Hall 1994), which was the area used in our calculations. Because the ecological impacts of these gears are comparable, the spatial extent and fishing intensity of each were calculated separately and subsequently combined as 'scallop dredges'.

To account for the active region of a benthic trawl, we used 24 m as the width for calculating the swept area of an otter trawl (Carrothers 1980, Hiddink et al. 2007). The area for *Nephrops* trawls was taken from Hinz et al. (2008), whose side-scan sonar observations showed tracks up to 60 m wide in the northeast Irish Sea. These gears were combined and are presented as 'otter trawls'.

The locations and swept areas for the different bottom-fishing gears were calculated for both 2006 and 2007. A chi-squared test was conducted to analyse whether the particular gears deployed in cells in 2006 tended to be the same as those in 2007.

**Fishing distribution and intensity.** The exact tracks of fishing vessels are not recorded by VMS, and previous studies have either analysed VMS records as point data (Rijnsdorp et al. 1998) or estimated vessel tracks between points (Eastwood et al. 2007, Mills et al. 2007). We used point data in this study and calculated the spatial extent of the potential surface area fished around each point for 2006 and 2007. The extent was estimated from each VMS record based on vessel speed, VMS interval and width of fishing gear. For example, for a beam trawl 24 m wide fishing at 4.5 knots

($8.334$ km h$^{-1}$), 1 VMS record (2 h interval) corresponds to a fished surface area of $0.024 \times 8.334 \times 2 = 0.4$ km$^2$.

Trawling effort derived from VMS records may be represented using high-resolution grids, with cell sizes of <3 km$^2$ (Mills et al. 2007) and 1 n mile$^2$ or 1 km$^2$ having been used for fishing effort analyses (e.g. Rijnsdorp et al. 1998, Hinz et al. 2009). For this study, we chose to use 1 km$^2$ cells as an appropriate compromise between the high-resolution VMS point data and the uncertainties in the calculations of surface area fished around each point. Gridded intensity of fishing activity for each year was calculated from the annual number of trawls estimated to have passed through each cell and their associated fished surface area. An intensity score of 0.4 therefore represented a fished area of 0.4 km$^2$ yr$^{-1}$. A fished intensity of 1 could describe a situation where the entire area of a 1 km$^2$ cell was fished once a year, assuming the trawl distribution was homogeneous. Intensity scores >1 indicate that a cell or parts of a cell were fished more than once annually. To test the similarity of fishing intensity per cell in 2006 and 2007, we calculated Pearson correlation coefficients.

To identify the spatio-temporal distribution of bottom-fishing effort by habitat type, VMS records were spatially joined to British Geological Survey (BGS) sediment type data using the ArcGIS geographical information system (GIS; ESRI). This linked bottom-fishing effort to the sediment classification scheme of Folk (1954). These classes were grouped into 5 habitat types, based on the largest proportion of constituent particle size in the BGS classification scheme. Mud, muddy sand, sand, gravel and reef are the habitat descriptors widely used in studies of bottom-fishing impact on the benthos (e.g. Collie et al. 2000, Kaiser et al. 2006, Pitcher et al. 2009). Biogenic habitats constructed or composed primarily of living biota are classed as rock in the BGS system. The potential for recovery of the seabed in UK waters following bottom-fishing was then calculated using published recovery rates of each habitat type, based on the distribution and intensity of fishing activity.

**Recovery from benthic fishing.** Published studies have found that the effects of bottom-fishing on habitats were most strongly related to types of gear and sediment, with sediment type itself correlated with depth (Kaiser et al. 2006, Pitcher et al. 2009). Natural disturbance, such as that caused by near-bed currents and wind-induced waves, is also related to depth and so it is implicit in habitat type and recovery. Therefore, habitats are generally defined in terms of their particle-size ranges, and this approach was adopted here. A review was conducted of scientific literature for recovery rates of the benthos in different habitats after bottom-fishing by the 3 main benthic gear types. Studies were identified using computer database search engines of peer-reviewed literature, such as Scopus and Aquatic Sciences and Fisheries Abstracts (ASFA), and general internet search engines. In our study area, primary and secondary production are large with strong seasonal patterns, so data from studies conducted in this, or similar, areas were used. The review by Kaiser et al. (2006) provided the majority of the recovery periods, and the results of 4 additional studies were used where particular habitat–gear combinations were not given in that review.

We used the term recovery as defined by Kaiser et al. (2006); recovery was deemed to have occurred when the abundance, species richness or biomass of benthic biota was equivalent to a 20% reduction or less in the pre-impact value. The study area has a long history of high levels of fishing activity, with trawlers tending to repeatedly target the same grounds year after year (Kaiser et al. 2002, Hiddink et al. 2006b). Therefore, the point at which recovery is deemed to have occurred is, for some habitats, a point in a constant disturbance cycle, and not disturbance of a pristine benthic community.

An index of recovery ($Ind_{\mathrm{Rec}}$) was calculated to estimate whether the frequency of bottom-fishing allowed habitats enough time to recover between fishing events. The fishing intensity scores (see above) were used to estimate the mean number of days between fishing events, per cell, per year. For example, a fishing intensity score of 0.4 represents 912.5 d between sweeps by fishing gear. To provide an $Ind_{\mathrm{Rec}}$ for each 1 km$^2$ cell, the recovery periods established from the literature review were divided by the mean number of days between fishing events. Where more than 1 gear was deployed in a cell, the longest recovery period was used for the portion of the cell affected by that gear. Where $Ind_{\mathrm{Rec}} = 1$, fishing intensity was equivalent to the required number of days for a habitat to recover; >1, the habitat was unable to fully recover; <1, recovery can be expected before the next fishing event. Benthic fishing fleets are active throughout the year in UK waters, although effort was noticeably reduced in December. The mean ± standard error of the mean (SEM) sizes of fished area per month by gear type, excluding December, were: beam trawling, 8375 ± 322 km$^2$; otter trawling, 7331 ± 279 km$^2$; and scallop dredging, 279 ± 49 km$^2$. During December, effort was between 10 and 30% of the other 11 months. Given the consistency of fishing activity, we assumed a homogeneous distribution of pressure during the year. This also facilitated comparison of fishing effort with aggregate extraction records, the latter only being available as annual data.

**Cumulative impact assessment.** To investigate cumulative impacts of aggregate extraction and fishing pressures, we first identified where they were coincident. Electronic monitoring systems (EMS) have been fitted to all aggregate dredgers operating in UK (England and Wales) waters since 1993 to automatically record position and 'dredging status' every 30 s. Aggregate extraction effort in the UK is measured as hours dredged annually (h $yr^{-1}$). EMS data provided in $50 \times 50$ m (2500 $m^2$) cells were analysed by Foden et al. (2009). We spatially combined 2007 EMS data and the 2007 gridded 1 $km^2$ fishing cells. The locations and habitats where both pressures coincided could then be identified. We cross-referenced the sediment descriptors from Foden et al. (2009) and associated them with the habitat types used herein for recovery from benthic fishing (Table 1). Where the pressures were concurrent in 2007, the recovery times of each habitat type from aggregate extraction and fishing were estimated. For recovery from aggregate extraction ($T_{BioAgg}$) we used values from Foden et al. (2009); recovery was defined as the mean time necessary for establishment of a benthic community virtually indistinguishable from surrounding, non-impacted reference sites (Cooper et al. 2005), or when species number, abundance and/or biomass of a site had returned to 90% of the original values (Hiddink et al. 2006b). Recovery from fishing was estimated for each habitat–gear combination as the mean fishing intensity in 2007 × recovery period (from Table 4).

To quantify cumulative impacts where benthic fishing occurred in active aggregate extraction sites, we estimated impact as the total recovery times under 4 scenarios. The premise for Scenario I was that the single worst or dominant pressure (i.e. the pressure requiring the greatest period for the benthos to recover) takes precedence over the others in determining combined effects, and lesser pressures have no additional impact (Bruland et al. 1991). Previous investigations of the sensitivity of marine landscapes to a range of human activities have found sand, coarse and mixed sediment environments to be more sensitive to extraction pressures (aggregate removal) than to abrasion pressures (benthic fishing; Stelzenmüller et al. 2010). Therefore, aggregate extraction was considered to be the dominant pressure. For Scenario II, multiple pressures were assumed to act independently within the system, and therefore overall recovery time was the sum of both aggregate extraction and benthic fishing (e.g. Crain et al. 2008, Halpern et al. 2008a).

The purpose of Scenarios III and IV was to show a range in the sensitivity of habitats to impacts that interact. Scenario III estimated cumulative impacts as the antagonistic effects of multiple pressures (Darling & Côté 2008); if pressures are applied consecutively to marine habitats, then the impact of the first pressure may precondition the habitat to be less sensitive to the second pressure. The total recovery period for Scenario III was estimated as recovery time from the primary pressure + 50% recovery time from the secondary pressure. Recovery times were expected to be between those of Scenarios I and II. In Scenario IV, synergistic effects were assumed, in which the impact from accumulated pressures was greater than the sum of the individual parts (Cefas 2001, Oakwood Environmental 2002), the assumption being that the first pressure lessens the resilience of a habitat, making it more sensitive to subsequent pressures. Therefore, in Scenario IV, estimated total recovery time was recovery from the primary pressure + 150% recovery time from secondary pressure, with the expectation that the total recovery times will be greater than in the other 3 scenarios.

Table 1. British Geological Survey (BGS) sediments (sensu Folk 1954) affected by bottom-fishing activity in UK waters in 2006 and 2007, showing mean and SEM of intensity of fishing per year. Sediments were grouped to habitat types for this study

| BGS sediment type | Habitat | Intensity (times fished $yr^{-1}$) 2006 Mean | 2006 SEM | 2007 Mean | 2007 SEM |
|---|---|---|---|---|---|
| Sand<br>Gravelly sand<br>Slightly gravelly sand | Sand | 1.14 | 0.005 | 1.12 | 0.005 |
| Gravel<br>Muddy gravel<br>Muddy sandy gravel<br>Sandy gravel | Gravel | 1.75 | 0.018 | 1.46 | 0.012 |
| Muddy sand<br>Sandy mud<br>Gravelly mud<br>Gravelly muddy sand<br>Slightly gravelly muddy sand | Muddy sand | 2.3 | 0.021 | 1.92 | 0.019 |
| Rock | Reef (biogenic) | 1.72 | 0.122 | 1.77 | 0.128 |
| Mud<br>Slightly gravelly mud<br>Slightly gravelly sandy mud | Mud | 4.67 | 0.073 | 6.91 | 0.122 |

## RESULTS

### Fishing distribution and intensity

In 2006, 122 000 $km^2$ (47.2%) of the English and Welsh seabed was estimated to be affected by benthic fishing, increasing in area to 134 000 $km^2$ (51.8%) in 2007. Fishing activity was patchily distributed, with strong spatial variation in annual fishing intensity in UK waters. In locations where towed bottom-fishing was recorded by VMS, intensity scores ranged from 0.0002 to

39 in 2006 (i.e. a 1 $km^2$ cell may have been fished up to 39 times $yr^{-1}$) and 0.0002 to 30 in 2007. In 2006, 42% of the UK's bottom-fished area was fished more frequently than once a year; this value decreased slightly to 40% in 2007.

There was a strong spatio-temporal correlation in fishing intensity between 2006 and 2007 (Pearson's $r = 0.405$, $p < 0.001$), i.e. the spatial pattern of fishing intensity remained broadly uncharged. In particular, there was a consistent pattern of intense activity near the northeast coast of England, and low or no activity to the north and west of Anglesey in the Irish Sea. As the spatio-temporal patterns of fishing pressure in UK waters have been mapped previously by Stelzenmüller et al. (2008), we have not reproduced the data here.

The spatial distribution of bottom-fishing coincided with 16 sediment classes grouped into 5 habitats (Table 1). Mean fishing intensity per habitat type ranged between 1.1 and 6.9 $yr^{-1}$. In relation to total habitat size, mud was the most intensively fished habitat in both 2006 and 2007. In terms of absolute fished area, the habitats most commonly targeted for bottom-fishing were sand and gravel (Table 2). These constitute relatively large habitats in UK waters (~226 000 $km^2$ in total), and in 2006 and 2007, fishing affected approximately 50% of their areas. In contrast, mud is the smallest habitat of the 5 (~2000 $km^2$) in UK waters, but 77% of its area was bottom-fished in 2006, increasing to 80% in 2007. A large proportion (63% in 2006 and 76% in 2007) of muddy sand was bottom-fished, but only 12% of reef habitat in 2006 and 2007.

Fishing gears were deployed in different proportions in each habitat (Table 3). However, there was a strong spatial overlap in the locations where particular gears were deployed in 2006 and 2007 ($\chi^2 = 63\,766$, $p < 0.001$). Scallop dredgers were the gear deployed in the smallest proportions of all habitats, ranging from 0.4% of mud to 11% of reef habitats.

## Recovery from benthic fishing

Table 4 summarises the recovery periods of 5 habitats after fishing by 3 types of benthic gears. Many studies did not precisely state a recovery time, rather indications were given such as >8 yr or <6 mo. In the absence of exact values, we ignored these inequality symbols and converted all recovery periods to number of days. Where a range of recovery times was available for a habitat–gear combination, we took the longest stated time. No published data were found for recovery rates following beam trawling in gravel, biogenic reef and mud habitats, nor from scallop dredging in mud habitats. However, these 4 habitat–gear combinations only represent approximately 1% of the total area subject to bottom-fishing in 2007. For all the other habitat–gear combinations, the recovery times in Table 4 were used to estimate the proportion of each habitat able to recover or not recover (Table 5). As there were no statistically significant spatial or temporal differences in gear

Table 2. Seabed habitats subjected to bottom-fishing in UK waters, showing total habitat size, mean areas bottom-fished and unfished and the proportion (%) of habitat fished in 2006 and 2007

| | Habitat type | | | | |
|---|---|---|---|---|---|
| | Sand | Gravel | Muddy sand | Reef | Mud |
| Total habitat size ($km^2$) | 185817 | 40041 | 23722 | 5085 | 1843 |
| Unfished area ($km^2$) | 94139 | 22264 | 7181 | 4458 | 394 |
| Fished area ($km^2$) | 91677 | 17776 | 16541 | 627 | 1448 |
| Habitat fished (%) | 49.3 | 44.4 | 69.7 | 12.3 | 78.6 |

Table 4. Recovery times (d) for habitats by fishing gear types. nd: no data

| Gear type | Habitat | | | | |
|---|---|---|---|---|---|
| | Sand | Gravel | Muddy sand | Reef | Mud |
| Beam trawl | 182[a] | nd | 236[b] | nd | nd |
| Otter trawl | 0[b] | 365[d] | 213[c] | 2922[b] | 8[b] |
| Scallop dredge | 2922[b,e] | 2922[b] | 589[b] | 1175[b] | nd |

[a]Kaiser et al. (1998); [b]Kaiser et al. (2006); [c]Ragnarsson & Lindegarth (2009); [d]Kenchington et al. (2006); [e]Gilkinson et al. (2005)

Table 3. Proportions (%) of the bottom-fished area by gear type in 2006 and 2007

| | Habitat type | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Sand | | Gravel | | Muddy sand | | Reef | | Mud | |
| | 2006 | 2007 | 2006 | 2007 | 2006 | 2007 | 2006 | 2007 | 2006 | 2007 |
| Dredge only | 2.5 | 1.7 | 8.2 | 8.1 | 3.4 | 2.3 | 8.7 | 11.0 | 0.7 | 0.4 |
| Beam only | 41.7 | 28.8 | 24.3 | 8.2 | 13.5 | 6.7 | 42.0 | 22.7 | 1.2 | 0.1 |
| Otter & *Nephrops* only | 31.9 | 41.5 | 33.6 | 40.4 | 64.0 | 76.8 | 30.8 | 34.2 | 93.5 | 97.9 |
| >1 gear type | 23.9 | 28.0 | 33.9 | 43.3 | 19.1 | 14.2 | 18.5 | 32.1 | 4.6 | 1.6 |

Table 5. Percentage (area, $km^2$) of habitats estimated to recover / not recover from 2007 levels of fishing intensity based on recovery times in Table 4

| Recovery status | Habitat: Sand | Gravel | Muddy sand | Reef | Mud |
|---|---|---|---|---|---|
| Recovered | 89.9 (84485) | 50.3 (10005) | 63.1 (11412) | 20.0 (129) | 99.5 (1469) |
| Not recovered | 10.1 (9462) | 41.5 (8263) | 36.9 (6676) | 57.4 (371) | 0.0 (0) |
| Undetermined | 0.0 (0) | 8.2[a] (1 626) | 0.0 (0) | 22.7[a] (147) | 0.5[a,b] (8) |

[a]No recovery period for beam trawl; [b]No recovery period for scallop dredge

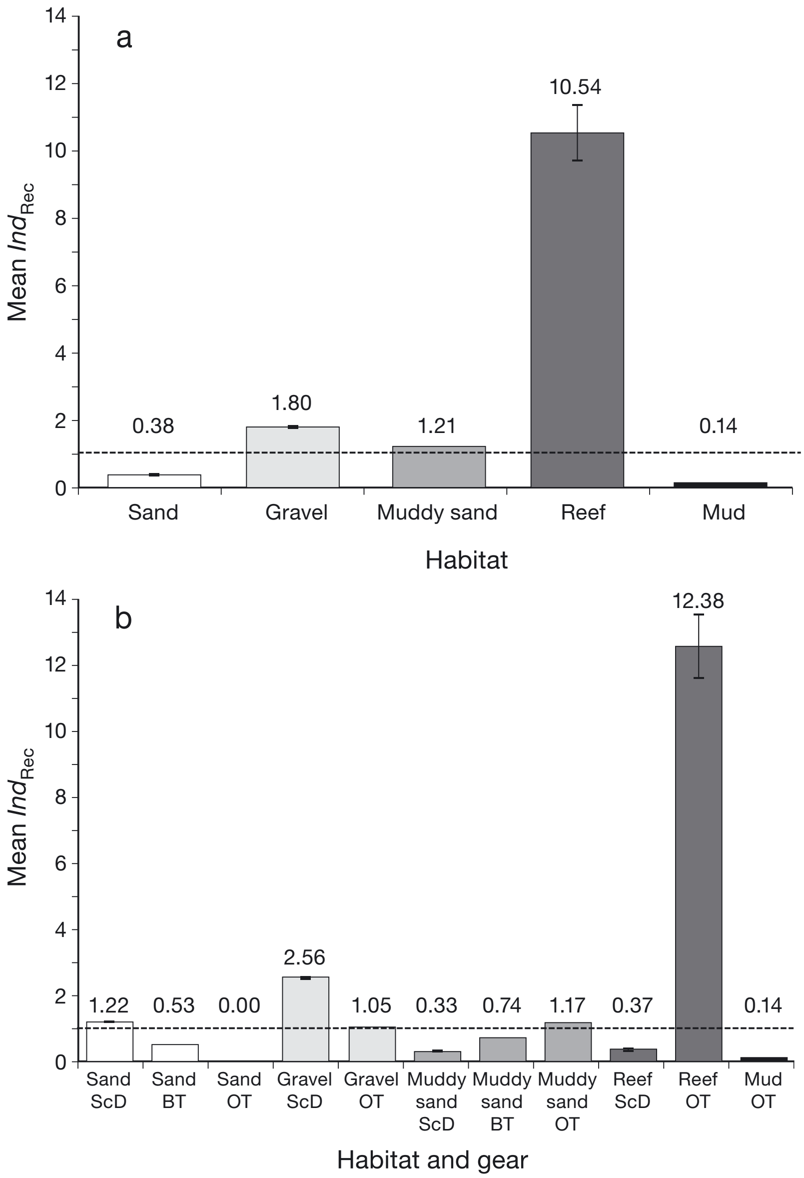


Fig. 1. Mean index of recovery ($Ind_{Rec}$) in 2007 for (a) habitats regardless of gear type and (b) gear–habitat combinations. BT: beam trawl; OT: otter trawl; ScD: scallop dredge. Error bars show SEM. At $Ind_{Rec}$ = 1, the recovery period equals fishing frequency (horizontal broken line on each plot); at <1, fishing frequency is less than the predicted recovery period; and at >1, fishing frequency exceeds the recovery period

types or fishing intensity between 2006 and 2007, we estimated recovery at 2007 fishing pressure levels. Values in Table 4 were also used to estimate $Ind_{Rec}$ for each 1 $km^2$ cell. The results are presented in Fig. 1 as a mean $Ind_{Rec}$ for each habitat and as mean $Ind_{Rec}$ for each habitat–gear combination.

In 2007, at an average fishing intensity for gear types, sand and mud habitats appeared to be able to fully recover, whereas gravel, muddy sand and reef habitats were fished at frequencies in excess of estimated recovery periods (Fig. 1a). However, $Ind_{Rec}$ (Fig. 1b) varied significantly with individual habitat–gear combinations. $Ind_{Rec}$ was less than 1 in 6 habitat–gear combinations and was ~1 for otter trawling in gravel. High fishing intensities in 4 habitat–gear types were responsible for $Ind_{Rec}$ exceeding 1. On average, scallop dredgers were deployed in sand too often to allow the habitat to recover. Scallop dredgers affected habitats 1.22 times more frequently than the estimated period required for recovery between sweeps. Beam and otter trawls are more commonly deployed gears in sand habitats (Table 3) and have an $Ind_{Rec}$ less than 1, resulting in a mean $Ind_{Rec}$ of 0.38 for the whole habitat. Scallop dredgers in gravel and otter trawls in muddy sand and reefs were also fished at frequencies unlikely to allow sufficient time for the habitat to recover.

Recovery was possible in 90% or more of the bottom-fished area of sand and mud, decreasing to approximately two-thirds of the fished area of muddy sand and half of gravel habitats (Table 5). In reef habitat, 20% was

Table 6. Cumulative impacts of benthic fishing at aggregate (agg) extraction sites in the UK, in 2007. See 'Materials and Methods' for explanation of recovery times from fishing and aggregate extraction ($T_{BioAgg}$). Cumulative impacts for 4 scenarios: I: longest recovery period; II: additive recovery period (fishing recovery + $T_{BioAgg}$); III: antagonistic recovery period ($T_{BioAgg}$ + 50% fishing recovery); synergistic recovery period ($T_{BioAgg}$ + 150% fishing recovery). BT: beam trawl, OT: otter trawl, ScD: scallop dredge, nd: no data

| Habitat fishing gears in aggregate extraction sites | Area (km$^2$) | Recovery after fishing once yr$^{-1}$ (from Table 4) (d) | Mean fishing intensity (times fished yr$^{-1}$) | Estimated recovery | | Cumulative impact (d) SCENARIOS | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | Time after fishing (d) | Time after agg extraction (d) ($T_{BioAgg}$) | I | II | III | IV |
| Sand BT | 18.34 | 182 | 0.89 | 163 | 2666 | 2666 | 2829 | 2747 | 2911 |
| Sand OT | 1.45 | 0 | 0.38 | 0.0 | 2666 | 2666 | 2666 | 2666 | 2666 |
| Sand ScD | 0.55 | 2922 | 0.19 | 561 | 2666 | 2666 | 3227 | 2947 | 3508 |
| Gravel BT | 6.36 | nd | 0.62 | nd | 3287 | 3287 | 3287[a] | 3287[a] | 3287[a] |
| Gravel OT | 3.46 | 365 | 1.26 | 459 | 3287 | 3287 | 3746 | 3516 | 3976 |
| Gravel ScD | 9.99 | 2922 | 0.35 | 1017 | 3287 | 3287 | 4304 | 3795 | 4813 |

[a]No recovery period for beam trawling in gravel, therefore values are aggregate $T_{BioAgg}$ only

fished at low enough intensity to allow for recovery. For 147 km$^2$ of reef, no estimate could be made of whether recovery was possible.

### Cumulative impact assessment

Aggregate extraction took place in 135 km$^2$ of UK seabed in 2007, in 40 km$^2$ of which benthic fishing also occurred. The pressures were found to be concurrent in sand (20.3 km$^2$) and gravel (19.8 km$^2$), in which scallop dredges, beam and otter trawls were deployed. Table 6 shows the size of areas for particular habitat–fishing gear combinations in aggregate extraction sites. Benthic fishing in these locations occurred less than once a year in 5 habitat–gear combinations and greater than once a year for otter trawling in gravel. Therefore, recovery times from fishing were estimated in proportion to fishing intensity, from the values in Table 4. The mean ± SEM recovery times from aggregate extraction alone were estimated at 7.3 ± 2.39 yr in sand and 9.0 ± 2.1 yr in gravel.

The cumulative impact of aggregate extraction and benthic fishing was estimated from the number of days required for the seabed to recover from these pressures at current levels (Table 6). In Scenarios III and IV, the primary pressure was aggregate extraction and the secondary pressure was fishing, because sand and gravel habitats are more sensitive to the former. Under the 4 scenarios, total recovery times were estimated between 7.3 and 13.2 yr. The estimated $T_{BioAgg}$ from aggregate extraction dominated the cumulative recovery time calculations. Gravel habitat required longer periods of recovery than sand habitat from the cumulative impacts of aggregate extraction and benthic fishing, regardless of fishing gear. In gravel habitat, cumulative recovery time was greatest from scallop dredging and aggregate extraction under Scenarios II, III and most especially from the modelled synergistic effects in Scenario IV. Deployment of otter trawls in sand at aggregate extraction sites did not result in a cumulative effect of increased recovery time. The impact of beam trawls and aggregate extraction in gravel could not be estimated (Table 4).

## DISCUSSION

The objectives of this study were to calculate, at a fine resolution, the spatial extent and intensity of benthic fishing in UK waters, and to estimate the proportion of fished habitats in which recovery would be possible at 2007 levels of fishing effort. Furthermore, we intended to identify locations where benthic fishing and aggregate extraction coincided in order to conduct a cumulative impact assessment under the scenarios of greatest, additive and interactive (antagonistic and synergistic) effects.

### Fishing distribution and intensity

Benthic fishing is the most widespread pressure acting on the continental shelf, with half to three-quarters trawled annually (Watling & Norse 1998, Kaiser et al. 2002). In mapping the spatial extent and intensity of benthic fishing, we found that approximately half of the UK seabed was affected in 2006 and 2007. The estimated 134 000 km$^2$ total bottom-fished area in 2007 is considerably higher than some previous authors' estimates (e.g. Eastwood et al. 2007, Stelzenmüller et al. 2008). This is because we calculated the swept area of otter trawling from the entire gear width (i.e. 24 m; Hiddink et al. 2007), rather than only the 2 × 2 m trawl doors. Similarly, the area of impact for *Nephrops* trawls was calculated from a gear width of 60 m, as observed by Hinz et al. (2008).

The availability of high-resolution VMS data was fundamental to our study, but VMS records have 2 notable limitations. Firstly, point data do not record the exact tracks of fishing vessels. Calculating the spatial extent of the potential surface area fished around each point will inevitably include some locations that were not fished and exclude areas that were. Nevertheless, this has been an established approach used by others (e.g. Rijnsdorp et al. 1998). Secondly, we expect total fishing effort to be underestimated close to the coast. Vessels of ≥15 m are able to fish within 12 n miles, provided they adhere to local Marine and Fisheries Agency (MFA) and Sea Fisheries Committees restrictions, but the majority of their effort occurs in waters farther offshore. However, we were unable to consider the effort by vessels <15 m in length, fishing predominantly within 12 n miles, because they are not required to carry VMS. A project is underway to build a dataset of inshore fisheries effort from Sea Fisheries Committees' observations, and to compare and integrate this information with that available from vessels carrying VMS. Preliminary results suggest that fishing effort in inshore areas is comparable to levels of fishing effort in the adjacent offshore waters (K. Vanstaen pers. comm.). Further effort will be required to identify habitat-specific recovery rates for these data.

The landscapes identified as the most heavily bottom-fished are those representing soft seabed with weak or moderate tide stress (Stelzenmüller et al. 2008), as scallop dredges, beam and otter trawls are generally deployed in sediments where the chances of gear loss are small (Rijnsdorp et al. 1998, Piet et al. 2000). In keeping with these findings, we established that the majority of benthic fishing by area was in sand habitats. Fishing intensity varied considerably between habitats, being lowest in sand and highest in mud (Table 2). The spatial distribution of effort remained similar between 2006 and 2007, an annual consistency noted by Kaiser et al. (2002). Fishing intensity and inter-annual change in intensity between the 2 years were similar to those found by Stelzenmüller et al. (2008). These authors analysed spatio-temporal patterns of fishing pressure in UK waters since 2001 and found it to vary spatially by region, but within regions, patches of high fishing pressure remained centred at the same locations. Inevitably, fishing activity occurs where target species are most abundant (Swain & Wade 2003), resulting in a patchy distribution of fishing effort (Rijnsdorp et al. 1998, Stelzenmüller et al. 2008). This also helps explain why the deployment of specific gears broadly remains spatially unchanged over time, as fishing métiers are designed to target particular species.

**Recovery from benthic fishing**

The intensity of bottom-fishing was such that the seabed habitats in 80% of the area were estimated to have enough time to recover between fishing events. Bottom-fishing at frequencies greatly in excess of estimated recovery time in 19% of remaining habitats means benthic recovery is unlikely to be achieved. Recovery times could not be estimated from our literature review for beam trawling in mud, gravel and reef habitats, or for scallop dredging in mud. However, this was not a serious limitation as these habitat–gear combinations only accounted for approximately 1% of the entire fished area. Notable patterns were identifiable in recovery periods for the 5 different habitats, and for particular habitat–gear combinations.

The recovery time of habitats was influenced by gear width, gear penetration, fishing frequency and sediment grain size. The literature review showed that recovery periods generally increase with habitat stability or sediment grain size; soft sediment habitats recover more rapidly than hard (gravel or reef) habitats. When recovery times were coupled with fishing intensity data, recovery periods were longest in the hard habitats and shortest in soft. For example, otter trawls were the predominant gear deployed in mud, and benthic recovery after otter trawling is relatively rapid (i.e. days) in this environment (Sanchez et al. 2000, Kaiser et al. 2006). In contrast, recovery from benthic trawls may take several years in hard habitats (Kaiser et al. 2006, Kenchington et al. 2006). Scallop dredging in sand and gravel and otter trawling in muddy sand and reef habitats all occurred too frequently for the habitats to recover ($Ind_{Rec} > 1$), leaving the seabed area unable to recover from bottom-fishing. It is hoped this detailed information on the sensitivity of different habitats to particular gears will help inform marine managers when making marine planning decisions.

An important area for ongoing research is the chronic modification of benthic communities by long-term bottom-fishing (Jennings & Kaiser 1998, Kaiser et al. 2000, de Juan et al. 2007), which was not considered in our study. Infrequent, pulsed otter trawling has been found to allow fauna to recover more quickly with an overall lesser effect than more intense, chronic, long-term trawling (Henry et al. 2006, Hinz et al. 2009). However, continental shelves are heavily exploited, creating a dearth of historically unfished reference sites with similar environmental characteristics to fished sites (Watling & Norse 1998, Kaiser et al. 2002, de Juan et al. 2007). This may severely restrict the possibility of accurately quantifying both acute and chronic impacts of bottom-fishing across the range of seabed habitats.

## Cumulative impacts

Aggregate extraction in UK waters is spatially restricted but highly damaging in nature (e.g. Cooper 2005, Eastwood et al. 2007, Vanstaen et al. 2008, Foden et al. 2009, Stelzenmüller et al. 2010). In 2007, aggregate was extracted from an area of seabed <1% of that affected by benthic fishing. However, the impacts in terms of time required for the seabed to recover after cessation of extraction are more severe (Foden et al. 2009).

The fishing industry has expressed particular concern over the potential for cumulative effects in areas where there are local concentrations of marine aggregate extraction licences. Cooper (2005) noted that in general in the UK there is avoidance of areas licensed for aggregate extraction by static gear fishers, due to the potential for gear damage. We found that the spatial pattern of fishing effort in 2007 supports this finding: benthic fishing occurred in less than 30% of the area from which marine aggregate was extracted.

There has been little research quantifying the cumulative impacts of bottom-fishing and aggregate extraction on benthic communities and the wider ecosystem (Vanstaen et al. 2008), and our study is a contribution to this area of cumulative effects assessment. Although limited to only 2 pressures, the 4 scenarios of cumulative impacts provide useful insights of how benthic fishing and aggregate extraction in combination might affect the seabed. All scenarios showed that in site-specific activity on sand and gravel habitats, aggregate extraction appeared to be a greater pressure than benthic fishing, regardless of gear type. Estimated recovery from aggregate extraction ($T_{\text{BioAgg}}$) constituted 68 to 100% of the cumulative recovery time for each scenario, where calculations could be made for both pressures. The additive impact modelled in Scenario II may be the most appropriate for quantifying cumulative impact for a pair of pressures (Crain et al. 2008). Synergistic effects in Scenario IV estimated the longest recovery times for all habitat-fishing gear combinations in aggregate extraction sites. The weighting schemes of Scenarios III and IV (Stelzenmüller et al. 2010) should ideally encompass several more pressures, in which aggregate extraction and benthic fishing would be part of a rank order. It is hoped that future empirical data from observations or experiments will enable application of a more complex weighting scheme, making estimations of total impact more comprehensive.

Many other human activities in the marine environment are absent from this study, such as oil and gas extraction, renewable energy structures, dumping, pipelines and cables. Nevertheless, our findings contribute to increased understanding of differing marine habitat sensitivities, and the individual and cumulative impacts on habitats from 2 major human pressures. Further research into the impact of abrasion, obstruction, smothering and extraction pressures on the marine environment is underway, using spatial analyses similar to the methods herein. This will enable integrated assessments of cumulative impacts, which can begin to estimate the effects of the multitude of pressures acting simultaneously in many offshore locations.

*Acknowledgements.* We thank J. Lee at Cefas for support with VMS data and GIS analyses and 2 anonymous reviewers for their helpful comments, which improved the manuscript. This work was funded by a Natural Environment Research Council (NERC) studentship, a Cefas CASE award and by the Department for Environment Food and Rural Affairs (Defra) research contract AE1148.

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*Editorial responsibility: Jake Rice, Ottawa, Canada*

*Submitted: November 24, 2009; Accepted: May 14, 2010*
*Proofs received from author(s): July 15, 2010*