Vol. 586: 167–179, 2018
https://doi.org/10.3354/meps12392

MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser

Published January 11

Exploring habitat credits to manage the benthic impact in a mixed fishery

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ABSTRACT: The performance of a combined catch quota and habitat credit system was explored to manage the sustainable exploitation of a mix of demersal fish species and reduce the benthic impacts of bottom trawl fisheries using a dynamic state variable model approach. The model was parameterised for the Eastern English Channel demersal mixed fishery using otter trawls or dredges. Target species differed in their association with habitat types. Restricting catch quota for plaice and cod had a limited effect on benthic impact, except when reduced to very low values, forcing the vessels to stay in port. Quota management had a minimal influence on fishing behaviour and hence resulted in a minimal reduction of benthic impact. Habitat credits may reduce the benthic impacts of the trawl fisheries at a minimal loss of landings and revenue, as vessels are still able to reallocate their effort to less vulnerable fishing grounds, while allowing the fishery to catch their catch quota and maintain their revenue. Only if they are reduced to extremely low levels can habitat credits potentially constrain fishing activities to levels that prevent the fisheries from using up the catch quota for the target species.

KEY WORDS: Fleet dynamics · Dynamic state variable modelling · TAC · Total allowable catch · Mixed fisheries · Eastern English Channel · Plaice · Cod

INTRODUCTION

The impact of bottom trawl fisheries on benthic habitats and ecosystems is a matter of global concern (Kaiser et al. 2006, 2016). Bottom trawl fisheries use towed gears that make direct physical contact with the seafloor to catch species living or feeding close to the bottom substrate (Valdemarsen et al. 2007). As a consequence, these fisheries potentially reduce the structural complexity of habitats, injure or kill target and non-target species, alter species composition, and modify ecosystem processes such as benthic production (Dayton et al. 1995, Collie et al. 1997, Thrush et al. 1998, Kaiser et al. 2002).

The extent to which bottom trawl fisheries disturb benthic ecosystems depends on the type of fishing gear used and the frequency and distribution of fishing activities in an area (Kaiser et al. 2006, Rijnsdorp et al. 2015), but also depends on the sensitivity of habitats and benthic ecosystems to fishing disturbance (Collie et al. 2000, Hiddink et al. 2007, Eno et al. 2013). This sensitivity differs across habitats as a result of differences in natural disturbances (bottom shear stress, effects of waves), bottom typology (e.g. slope and depth), sediment composition, and species composition (Hall 1994, Jennings & Kaiser 1998, Snickars et al. 2014, van Denderen et al. 2014). Complex biogenic habitats with emergent structures are likely to be more affected than naturally disturbed, soft sedimentary habitats (Jennings & Kaiser 1998, Lindholm et al. 2015).

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Fish species caught by bottom trawl fisheries differ in their preference of habitat types because of their morphological and behavioural characteristics. Habitat preference generally changes throughout the life cycle of a species because of ontogenetic niche shifts (Thouzeau et al. 1991, Le Pape et al. 2003, Atkinson et al. 2004). Adult roundfish species, for example, have a preference for hard bottom habitats with high structural complexity which provide food and shelter (Tupper & Boutilier 1995, Wieland et al. 2009), while flatfish species such as plaice *Pleuronectes platessa* and sole *Solea solea* prefer soft bottom habitats in which they can bury themselves to avoid predation (Gibson & Robb 2000). As a result, the impact of a fishery on the seabed habitat and benthic ecosystem depends on the linkage between the habitat preference of the different species and the species preference of the fishery.

In many fisheries management systems around the world, management has historically focussed on the sustainable exploitation of commercial fish and shellfish stocks by setting total allowable catches (TACs) but ignoring the potential wider ecosystem effects of fishing (Pikitch et al. 2004, Holland & Schnier 2006, Chu 2009). Policy developments such as the amendments to the Magnuson-Stevens Act in the USA and the reform of the Common Fishery Policy (CFP) in Europe have recognised the importance of safeguarding ecosystem composition, structure, and functioning, and have embraced an ecosystem-based approach. This objective requires managers to ensure a sustainable use of natural resources while minimising the impacts of fishing activities on the structure and functioning of seabed habitats and benthic ecosystems. In this context, the EU has established marine Natura 2000 sites. While these sites do protect habitat features and species, this is not a fisheries management objective, but rather a conservation objective in relation to biodiversity.

The conventional approach of fisheries managers to maintaining or restoring ecosystem processes focusses on technical measures, including gear restrictions and spatial measures, as well as direct limitations on fleet capacity and fishing effort (Rice et al. 2012). Most spatial management measures aim to either protect habitat, preserve biodiversity, or maintain a reserve of fishes and often do not have a socio-economic management objective (Halpern 2003, Lester et al. 2009). As a result, socio-economic benefits of spatial management are not necessarily to be expected, and one could consider a spatial management measure to be successful as long as it meets its primary objectives without causing any socio-economic harm (Darling 2014, Caveen et al. 2015). Several studies have proposed the use of a credit system as an alternative management approach (Holland & Schnier 2006, Kraak et al. 2012). These credit systems try to balance economic and environmental values associated with fisheries by addressing specific conservation goals with limited effects on the fishery (Van Riel et al. 2015).

Our study explored the performance of individual catch quotas combined with a habitat credit system to manage the sustainable exploitation of a mix of demersal species and to minimise the benthic impacts of bottom trawl fisheries. Results were compared to traditional quota management for 2 commercial species, cod and plaice, which have different habitat associations. We applied an individual-based simulation model of the effort allocation and discarding decisions in a mixed fishery (Poos et al. 2010, Batsleer et al. 2013). The model was adapted to include individual habitat credits (IHC) and was parameterised for the French multipurpose bottom trawl fleet in the Eastern English Channel. This fleet can switch between dredging for scallops *Pecten maximus* and otter trawling for a mix of demersal fish species (Carpentier et al. 2009). We then evaluated the extent to which the incorporation of habitat credits in a quota management system can be a tool for ecosystem based management, in terms of benthic impact reductions and sustainable exploitation of fish.

MATERIALS AND METHODS

Study area

The Eastern English Channel (International Council for the Exploration of the Seas [ICES] division 7.d) consists of 15 ICES rectangles (~30 × 30 nautical miles [nmi]), which we divided into grid cells of 3 × 3 nmi each. Spatial distribution of habitats was derived from a detailed map of Eastern English Channel seabed habitats in Coggan & Diesing (2011) (Fig. 1). Each grid cell was assigned the dominant habitat type present within that cell. We distinguished 5 habitats based on grain size and depth (soft, infralittoral coarse, circalittoral coarse, deep coarse, and rock), following Coggan & Diesing (2011). Soft-sediment habitats, consisting of fine sand or muddy sediments, are found along the coast and in the eastern part where the English Channel borders the North Sea. Coarse sediment habitats are the predominant type in the Eastern English Channel. Rocky sea bed habitats consisting of a combination of

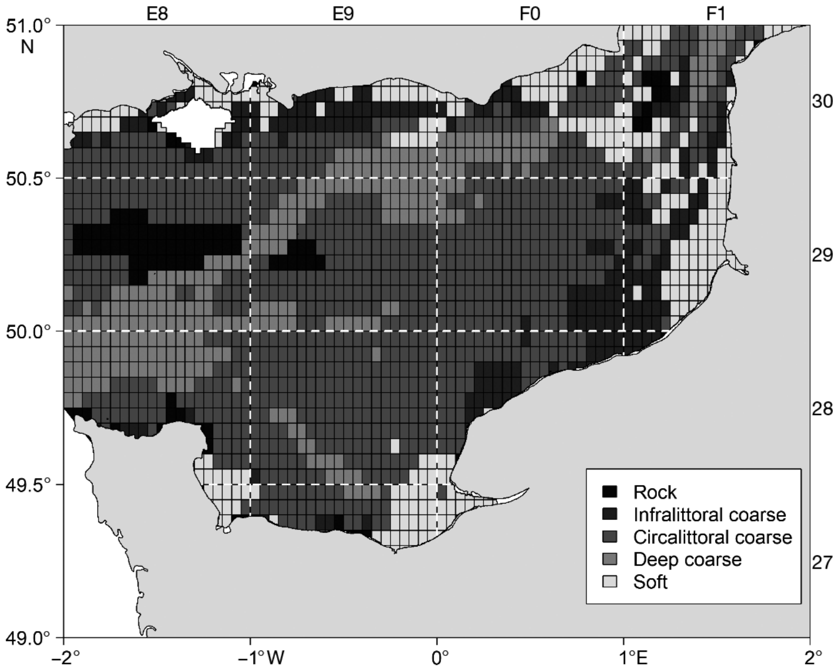

Fig. 1. Habitat distribution map based on Coggan & Diesing (2011), with the main French fishing harbours used in this study. ICES rectangles, defined by the letters and numerals along the top and right axes, are delineated by the white dashed lines

infralittoral and circalittoral rock and other hard substrates occur towards the west along the coast, with an extensive reef found in the western part of the study area (Diesing et al. 2009). Within each ICES rectangle, adjacent cells of the same habitat type were merged and counted as a single fishing area, resulting in 126 fishing areas.

Habitat impact credits were assigned to each fishing area based on the sensitivity of habitats to fishing activities. Rock habitat and deep coarse sediment habitat support emergent epibenthos such as sponges, bryozoans, and hydroids, making them more vulnerable to trawling than more shallow and dynamic soft sedimentary habitats (Jennings & Kaiser 1998, Kaiser et al. 2006, Eno et al. 2013, Boulcott et al. 2014). Because of the increasing sensitivity of habitats to fishing activities, impact credits were divided into 3 classes with increasing credits: infralittoral coarse and soft sediment habitat were assigned 2 credits, circalittoral coarse habitat was assigned 5 credits, and deep coarse and rock habitats were assigned 10 credits (Table 1). The classification of infralittoral coarse habitat as being less sensitive to fishing than circalittoral course and deep coarse habitat reflects the natural tidal disturbance of the sea bottom in the infralittoral, making its ecological community more resilient to additional disturbance from bottom fishing (van Denderen et al. 2015). If used in a policy setting, the impact credit allocated assigned to each habitat type should be decided by policy makers, and can be based on sensitivity matrices (Eno et al. 2013).

Study fleet

We selected the French multipurpose bottom trawl fleet as a case study because it uses bottom contact gears that affect bottom habitats. Vessels within this fleet switch between demersal otter trawling (OTB) and dredging for scallops *Pecten maximus*, making daily trips to their fishing grounds while operating from several ports around the Eastern English Channel, mainly Boulogne-Sur-Mer and Dieppe. The scallop fishery is fully open between November and April (Guyader et al. 2004). OTB is generally operated with 80 mm mesh size nets, targeting a variety of demersal species. The scallop dredge (DRB) lands few other commercially valuable species (Carpentier et al. 2009). The southern part of the Eastern English Channel is the most important scallop fishing ground, and some fishers temporarily change harbours so as to reduce their journey time to the scallop fishing grounds.

Catch data were derived from logbooks and sales slip data of the fleet for the period 2006 to 2010. The data included information on individual vessels by fishing trip on vessel length, engine power, gear type, mesh size, fishing grounds (ICES rectangle), fishing effort (hours fished), and the weight and value of the landings per species. Aggregated fishing effort estimates per fishing area were derived at a fine scale resolution (3 × 3 nmi) for 2008 from Delavenne (2012) (Table 1).

Table 1. Habitat types, associated habitat impact credits and fishing effort distribution (%) for demersal otter trawls (OTB) and scallop dredgers (DRB) estimated from Delavenne (2012)

Habitat	Habitat impact credit	Effort OTB (%)	Effort DRB (%)
Rock	10	1.0	0.2
Deep coarse	10	14.9	16.4
Circalittoral coarse	5	68.4	71.3
Infralittoral coarse	2	9.2	5.7
Soft	2	6.6	6.4
Total effort (h yr^{-1})		394 385	46 545

Spatial and temporal shifts in catch rate

A total of 5 species (scallops *P. maximus*, sole *Solea solea*, plaice *Pleuronectes platessa*, cod *Gadus morhua*, sea bass *Dicentrarchus labrax*) and 1 taxonomic grouping of cephalopods (which included cuttlefish *Sepia officinalis* and squid *Loligo vulgaris* and *L. forbesii*) were included in the analyses. Throughout the paper, these will be referred to as 6 'species', for the sake of simplicity, keeping in mind that one of these 'species' really represents the taxonomic grouping of cephalopods. These species were considered because of their economic importance for the multipurpose trawling fleet, i.e. combined, they contribute 67% of the total annual income.

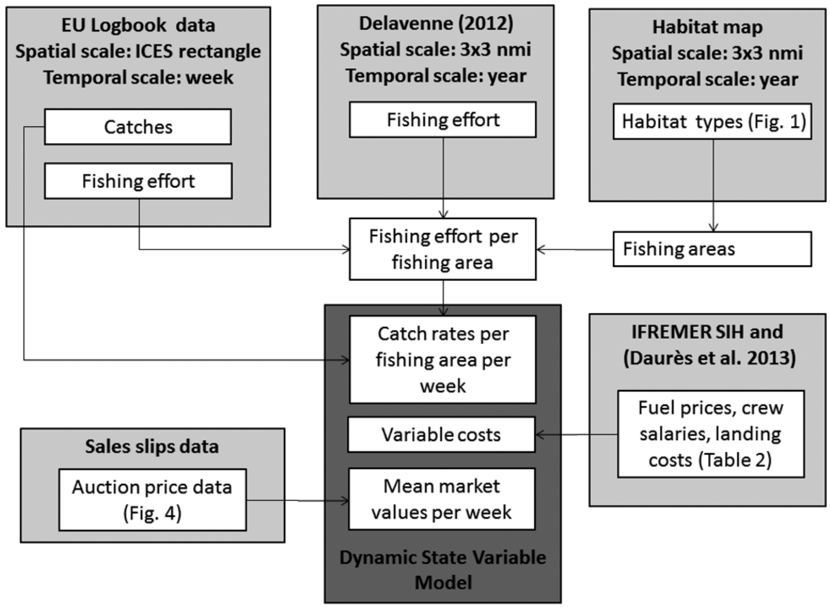

Fig. 2. Schematic representation of the methodology and data used in this study

Catches in the present study were estimated on the basis of landings per unit effort of French commercial vessels. For each species, the mean and variance of catches were estimated from the logbooks at their spatial (ICES rectangle) and temporal (week) scale (Fig. 2). These means and variance in catches (kg) were estimated using generalised additive models (GAMs) (Wood 2006). In the GAMs, fishing effort (hours) was used as model offset (Zuur 2012), allowing the prediction of catch rates (kg h^{-1}) from the GAM results. A negative binomial response with a logarithmic link function was applied (Zuur et al. 2009). Scallop catch rates were analysed only including trips with dredges, while catch rates of the other 5 species were analysed for trips operated by demersal otter trawls. The model estimating the catch C per species s by area per day is given by:

$$C_s = \alpha + \beta(\text{year}) + f_1(\text{engine power}) + f_2(\text{mesh size}) + f_3(\text{lat,lon,DOY}) + \log(\text{effort}) \quad (1)$$

where α is the model intercept and f_1, f_2, and f_3 are smooth functions based on a tensor product smoother (Zuur 2012). The differences in catch rates among years is estimated in the parameter β(year). The tensor product smoother f_1(engine power) was included due to its influence on the catch efficiency (Rijnsdorp et al. 2006), and f_2(mesh size) was included as the choice of mesh size may indicate the species the fishery is targeting. The interaction term f_3(lat,lon,DOY) fits the effects of space (latitude and longitude based on the geographic midpoint of the ICES rectangle) and day of year when the vessel returned to port (DOY). The term is based on a thin-plate spline (Wood 2003) for the space variables and a cyclic cubic regression spline for DOY. The cyclic cubic regression spline was chosen to receive equal values and slopes at the beginning and end of the year (Wood 2006). For scallops, the cyclic cubic regression spline for DOY was changed into a cubic regression spline because of the closed season from May to October. Akaike's information criterion (AIC) was used to assess the model fit, whereby the model with the lowest AIC value was selected as the best candidate. The best models included a limited degree of freedom for the f_1 and f_2 smoothing terms (k = 4), but a higher degree of freedom for the f_3 smoothing term of the space × time interaction (k = 9). All analyses were done using the 'mcgv' package within the R statistical program (version 2.12.1; R Core Team 2013, Wood 2006).

Catch rates by fishing area

The GAM provides estimates of catch rates per species per ICES rectangle. The rectangles are composed of mosaics of habitats (Fig. 1). To estimate catch rates by fishing area within ICES rectangles, catch rates were assumed to be positively correlated with effort per fishing area (Fig. 2). This assumption is supported by observations that fishing effort is not homogeneously distributed within an ICES rectangle, but is distributed on a finer spatial scale, concentrating within areas of high fish densities (Rijnsdorp et al. 2011).

Observed fishing effort per fishing area on the fine scale resolution (3 × 3 nmi) was derived from combining weekly effort from logbooks by ICES rectangle with fine-scale effort distribution (Table 1) as follows: First, the fraction of effort distributed over the habitat types within each ICES rectangle was calculated for each gear type based on Delavenne (2012). Multiplying these fractions with the effort distribution by ICES rectangle in the logbooks resulted in a distribution of effort E in habitat type h within an ICES rectangle a in week w (E_{haw}).

Catch rates at the resolution of the habitat type within each ICES rectangle (L_{ha}) were not available, because catch data are not collected at a scale within which different habitats can be detected. Meanwhile, information on the fine-scale distribution of fishing effort was available in Delavenne (2012). Although catch data in logbooks can be distributed over fishing positions recorded in vessel monitoring system (VMS) data, such procedures generally assume that catches are uniformly distributed over VMS positions (Dinmore et al. 2003, Gerritsen & Lordan 2011, Hintzen et al. 2012). However, fishing activities may concentrate within areas and habitat types with a high catch rates of target species (Rijnsdorp et al. 2011). To capture both assumptions of the relationship between effort and catch rates, we used the equation:

$$L_{ha} = \left\{ \frac{(L_a \times \sum E_{haw}) \times \left\{ \frac{E_{haw} \times \left(\frac{E_{haw}}{S_{ha}}\right)^{\alpha}}{\sum\left[E_{haw} \times \left(\frac{E_{haw}}{S_{ha}}\right)^{\alpha}\right]} \right\}}{E_{haw}} \right\} \quad (2)$$

where S_{ha} is the surface area of habitat type h in ICES rectangle a. The coefficient α is a scaling factor to aggregate the highest fish density within habitat types with the highest effort by area ratio. When α is set to 0, catch rates are equal over each habitat type within an ICES rectangle. Increasing α results in a differentiation in catch rates over habitat types, with larger catch rates in habitat types with the highest aggregation of effort (Fig. 3). The value of α was set to 0.5, thus assuming fishing activities concentrate

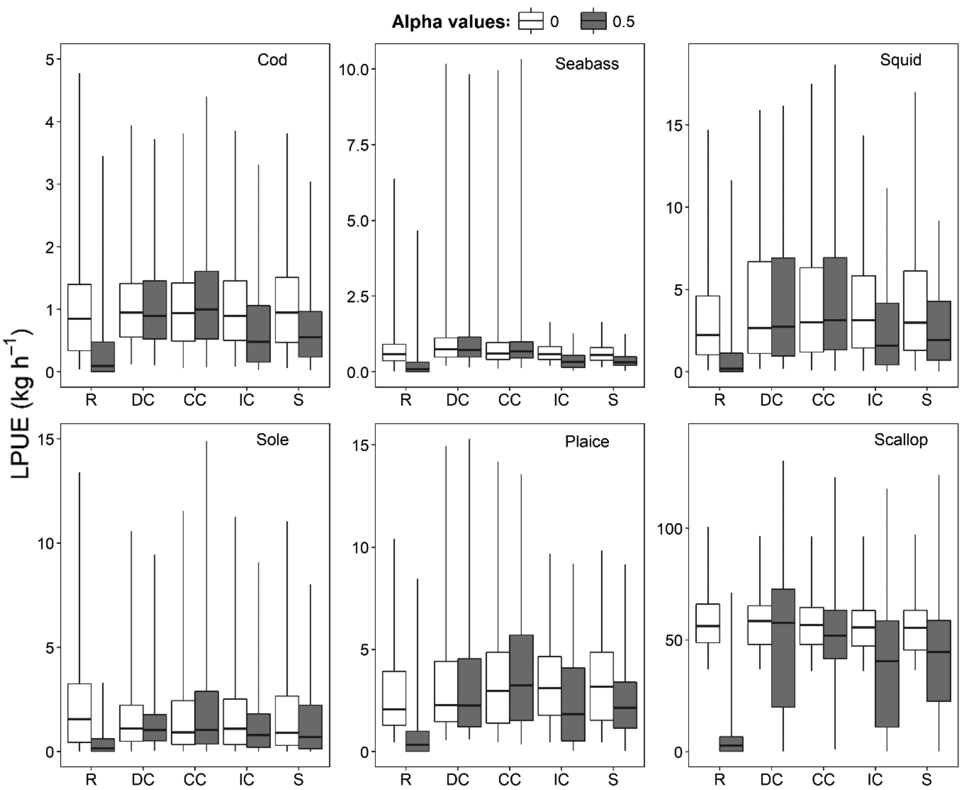

Fig. 3. Variation in catch rates (landings per unit effort [LPUE], kg h^{-1}) for 6 target species on different substrates. R: rock; DC: deep coarse sediment; CC: circalittoral coarse sediment; IC: infralittoral coarse sediment; S: soft sediment. Black horizontal bars correspond to medians, lower and upper hinges correspond to the first and third quartiles, whiskers extend to data range. The coefficient α is a scaling factor used to differentiate catch rates over habitat types. Setting α to zero results in equal catch rates over each habitat type, while increasing α results in larger catch rates in habitat types with the highest aggregation of effort

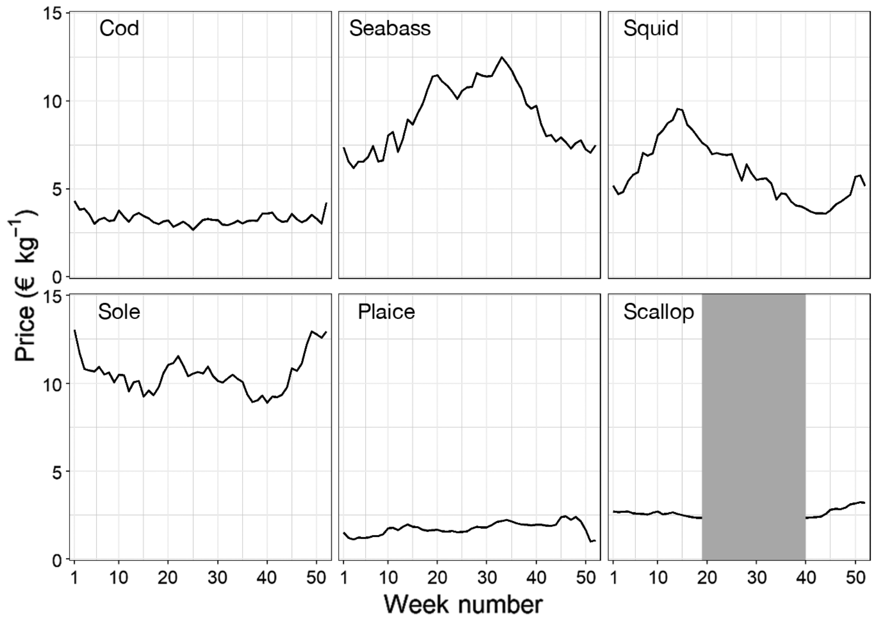

Fig. 4. Average ex-vessel price per week for each of the target species in the period 2006 to 2010. No prices are available for scallops from May to October because of the closed season (shaded area)

within areas and habitat types with high catch rates of target species (Rijnsdorp et al. 2011).

To scale the catch rates by week, estimates were rescaled to reflect the average fishing effort per week. Fishing effort per week was estimated from the logbooks, resulting in average time spent at sea (10 h d^{-1}) and average number of trips per week (4 trips wk^{-1}).

Economic data

Mean weekly market values for the 6 species were calculated from auction price data in the period from 2006 to 2010 (Fig. 4). During the scallop season, scallops contribute to more than 85% of the gross revenue and account for almost 40% of the annual income of the fleet (Carpentier et al. 2009). The 5 other species included in our study account for another 27% of the total annual income.

Table 2. Variable cost of the scallop dredgers (DRB) and demersal otter trawl (OTB) used in the simulation

Variable costs	OTB	DRB
Fuel (€ fishing d^{-1})	730	440
Crew costs (% of revenue)	40	40
Landing costs (% of revenue)	6	6
Other variable costs (€ fishing d^{-1})	40	40

The model includes a fine for overshooting the individual landings quota or IHC. The fines were set to values of €1950 kg^{-1} and €5000 IHC^{-1}, being several orders of magnitude higher than the market prices of the target species, ensuring vessels complied with the management regulations in the model.

Variable costs include fuel costs, landing costs, and crew salaries (Table 2). Fuel costs depend on the type of vessel and gear used. Fuel costs were approximately €440 fishing d^{-1} when operating a dredge and €730 fishing day^{-1} (F. Daurés pers. comm.) when operating a demersal otter trawl (in the period 2008 to 2010). Crew costs are 40% of the total revenue obtained at the end of the fishing trip. Landing costs are defined as the fees paid by the vessel when catches are landed and are 6% of the total revenue. Other variable costs (e.g. ice, bait, food) are €40 fishing day^{-1}. Landing costs, salary costs, and other variable costs were derived from a study by Daurès et al. (2013) and were assumed to be independent of the gear type used during a fishing trip.

Simulation model

The performance of a combined catch quota and habitat credit system was forecasted using a dynamic state variable model (DSVM) (Houston & McNamara 1999, Clark & Mangel 2000). Such models have been applied in fisheries research to forecast fishing strategies under different management and market constraints (Gillis et al. 1995, Poos et al. 2010, Dowling et al. 2012).

The model was parameterised for vessels from Boulogne-sur-Mer with the predicted spatial and temporal catch rates of the 6 species under consideration (scallops, sole, plaice, cod, sea bass, and cephalopods), ex-vessel prices (i.e. the price received at the point of landing the fish), and variable costs. Two gear types were modelled that could be used interchangeably: demersal otter trawls can be used throughout the year targeting the 4 finfish and cephalopod species, while scallop dredges can only be used during the scallop season. The model assumes individual skippers will maximise their expected annual net revenue by making weekly decisions on (1) whether to go fishing or not, (2) what type of gear to use, and (3) where to fish. These choices are affected by their annual catch quota and habitat credits.

The optimal fishing strategy in each week of the year, denoted by t, depends on the state of the individual skipper. In our case, the cumulative uptake of habitat credits as well as cumulative landings of the quota-constrained species affects the possibility of continuing to fish without exceeding the annual habitat credits or landing quotas. To simplify model results, simulations were done assuming that catch quota are restrictive for 2 species: cod and plaice. These species were chosen because they differ in their preference of habitat types (Tupper & Boutilier 1995, Gibson & Robb 2000).

DSVMs find the optimal behavioural strategy by working backwards, starting at the end of the year. The expected net revenue at the end of the year is linked to the choices in the preceding weeks by means of a value function between time t and the end of the year T. The value function for an individual skipper depends on the cumulative landings of the quota species L, the amount of quota U for cod or plaice, the cumulative uptake of habitat credits I, the amount of habitat credits available C, and the fine for exceeding the quota or habitat credits D, and is expressed as $V(L,U,I,C,D,t)$. Individuals exceeding their quota or habitat credits pay a fine that depends on the quota $(L - U)$ or credit $(I - C)$ overshoot and the fine per unit weight (d_1) and per credit (d_2). The state-dependent part of the revenue at the end of the year, after all fishing has been completed, $V(L,U,I,C,D,T)$, is defined by the fine of overshooting quota and/or credits; $\phi(L,U,I,C,D)$. $\phi(L,U,I,C,D)$ thus depends on the annual landings, uptake of habitat credits, quota, and available credits:

$$\Phi(L,U,I,C,D) = \begin{cases} 0, & L \le U \text{ and } I \le C \\ -(L-U)d_1, & L > U \text{ and } I \le C \\ -(I-C)d_2, & L \le U \text{ and } I > C \\ -(L-U)d_1 - (I-C)d_2, & L > U \text{ and } I > C \end{cases} \quad (3)$$

For each week before T, the expected net revenue is determined by the value function, the weekly gross revenue from the catch, and the cost of fishing. Each week, individuals choose to use a type of gear g, and to visit a fishing area a (including area 0: 'staying in harbour'). For all time t preceding T, we used stochastic dynamic programming to find the optimal solution by backward iteration of the net expected revenue H from t to the end of the year considering the choices a and g and the states L and I at t and optimal choices in subsequent weeks:

$$H(L,U,I,C,D,t;a,g) = R(a,g,t) \times \left(\tfrac{100}{K}\right) - C(a) + \mathbf{E}_{(a,g)}[V(L',U,I',C,D,t+1)] \quad (4)$$

where $R(a,g,t)$ is the expected immediate contribution of the gross revenue from the sales of the catch in a week resulting from choices a and g. The term κ represents the percentage that the 6 species contribute to the total income (i.e. 67%). The term $C(a)$ represents the variable costs in a week resulting from choosing fishing area a and using gear type g including fuel, crew, landing costs, and other variable costs. The term L' reflects the change of the state L resulting from the weekly landings for the quota species. This change is stochastic and depends on the means and variances found in the statistical analyses of the catch rates. The term I' reflects the change of state I as a result of the weekly choice of fishing in an area with a given amount of habitat credits. The term $\mathbf{E}_{(a,g)}[V(L',U,I',C,D,t+1)]$ denotes the expected value taken over all possible states resulting from choices a and g. The future utility given that an individual behaves optimally from time t onwards is:

$$V(L,U,I,C,D,t) = \max_{(a,g)}[H(L,U,I,C,D,t;a,g)] \quad (5)$$

Starting with $V(L,U,I,C,D,T) = \phi(L,U,I,C,F)$, we can iterate backwards in time and find the optimal choice in terms of location and gear type for all possible states, combining the direct net revenue obtained in a fishing trip and the effect of the fines when exceeding annual quota or habitat credits.

The expected direct gross revenue $R(a,g,t)$ from the sale of the 6 target species for any choice depends on the catch and the ex-vessel price in a given week. The catch is determined by the choice of gear and the spatial and temporal distribution of the target species.

Management scenarios

This study explored the performance of individual quota management for plaice and cod in combination with a habitat credit system to sustainably exploit resources and minimise the impact of the fishery on the benthic ecosystem. First, the performance of traditional quota management for plaice and cod was evaluated in relation to benthic impact. Individual quota for plaice were gradually increased from 1 to 8 tonnes (t) yr^{-1}. Given the low observed quotas for cod under the cod recovery plan (Kraak et al. 2013), individual quota for cod were lower, and increased

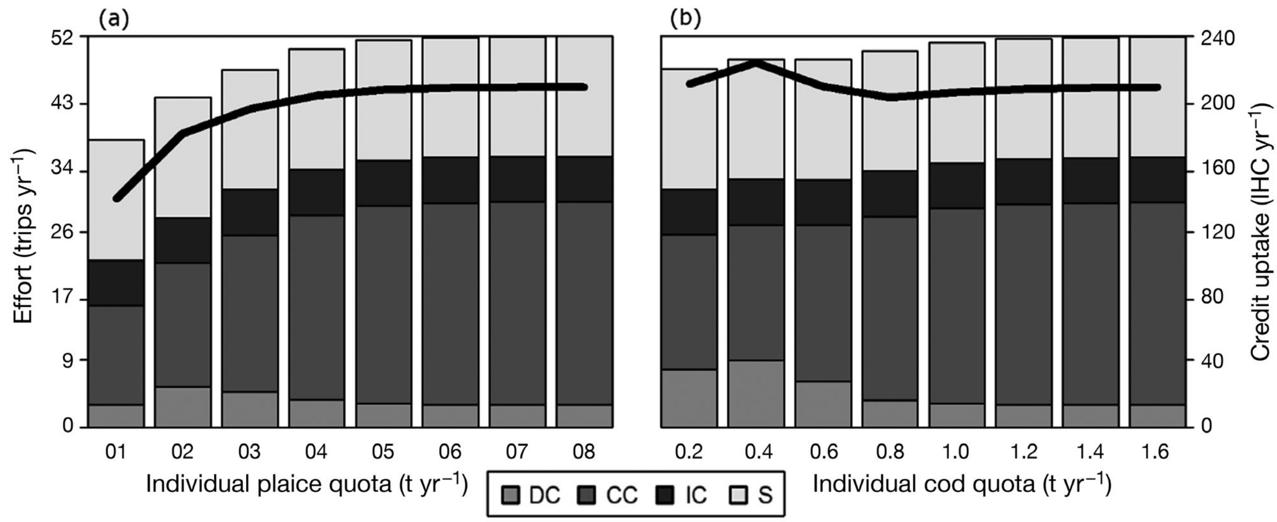

Fig. 5. Average number of trips in a year to a substrate type (DC: deep coarse; CC: circalittoral coarse; IC: infralittoral coarse; S: soft sediment), depending on the quota of (a) plaice and (b) cod. Black line: average impact (credit uptake). IHC: individual habitat credits

from 0.2 to 1.6 t yr^{-1}. Second, management scenarios were explored that combined individual catch quota and habitat credits. Habitat credits varied from 20 to 300 yr^{-1}, while quota varied from 1 to 9 t yr^{-1} and 0.1 to 1.5 t yr^{-1} for plaice and cod, respectively. The maximum of 300 habitat credits yr^{-1} was based on the maximum uptake of habitat credit by fishers in unconstrained model simulations.

RESULTS

Catch quota simulations

First, we explored the effect of individual catch quota on habitat impact by estimating the fishing effort in different habitat types. Reduction of the plaice quota affects the level of fishing effort and the choice of habitats (Fig. 5a). At high quota, vessels fish year-round (52 trips yr^{-1}), and the majority of trips occur in low impact, circalittoral coarse and soft sediment habitats. At plaice quotas below 4 t yr^{-1}, the number of trips is reduced, mainly in the circalittoral coarse sediment habitat. This effort reduction and reallocation results in a decrease in habitat impact over the year, measured as the use of habitat credits (Fig. 5a).

A reduction of the cod quota also affects the level of effort and the habitat choice (Fig. 5b). At the highest cod quota, a vessel will fish year-round, allocating the majority of trips over circalittoral coarse and soft sediment habitats. A decrease in the cod quota results in a gradual decrease to 48 trips yr^{-1} at the lowest cod quota and a slight increase in fishing in the more sensitive, deep coarse habitats. This reallocation results in a slight increase in habitat impact over the year in spite of the reduction in the number of trips (Fig. 5b).

Catch quota and habitat credit simulations

In this section, we explore how a habitat credit system can mitigate benthic impact in a fisheries managed by individual catch quota for the main target species. In this fishery, the amount of habitat credits allocated to the fishery affect the actual credit use up to 220 IHC yr^{-1}, but this depends on the quota, particularly for plaice (Fig. 6a,b).

When habitat credits are combined with plaice or cod quota, a decrease in habitat credits from 300 to 100 IHC has a limited influence on the overall effort of the fishery (Fig. 6c,d). At high IHC, effort will be allocated to both gear types, with most effort being allocated to the DRB (Fig. 6e,f). A decrease in IHC from 300 to 100 results in a reduction of OTB effort (Fig. 6g,h), but hardly affects DRB effort, irrespective of the level of the quotas. When IHC < 100, effort decreases and vessels no longer choose to fish using OTB, but only fish with the DRB (as long as IHC > 60).

Decreasing IHC from 300 to 100 only results in a modest (approximately 7%) reduction of net revenue (Fig. 6i,j). This is largely the result of a small increase in DRB effort that compensates for the reduction in OTB effort. When IHC are very low (<100), and the fishery is fully targeting scallops, net revenue sharply

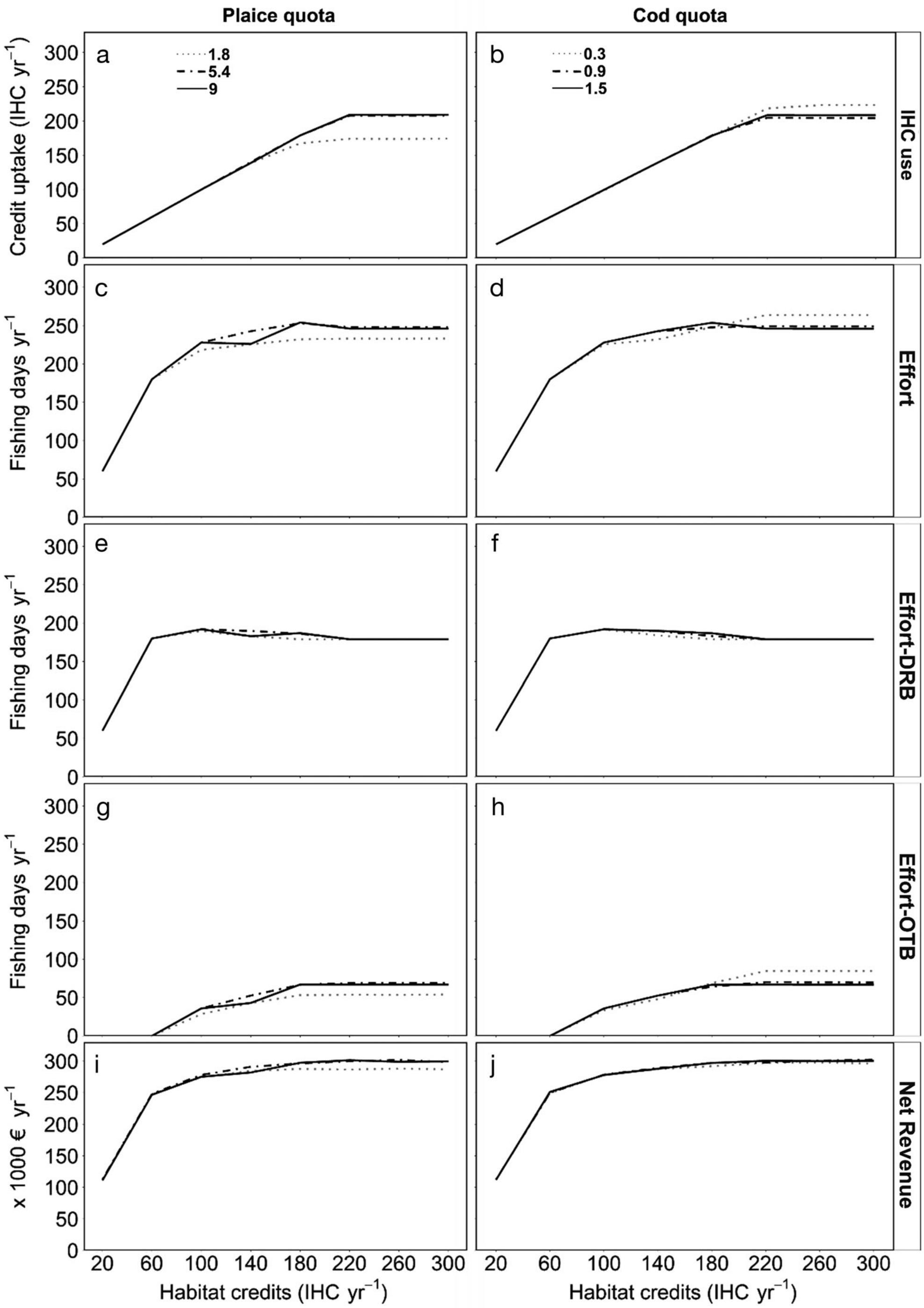

Fig. 6. Credit uptake, total effort, dredge effort, otter trawl effort, and net revenue in relation to the individual habitat credits (IHC) in combination with 3 levels of individual catch quota for plaice and cod. Plaice quota range from 1.8 to 9 t yr^{-1} and cod quota range from 0.3 to 1.5 t yr^{-1}. OTB: demersal otter trawling; DRB: scallop dredging

declines below €200 K. Because plaice and cod have a relatively lower economic value and a lower catch rate compared to other target species, the loss in revenue from plaice and cod at reduced catch quotas is compensated by increasing the DRB effort, targeting scallops. Our results thus suggest that when reducing habitat credits from high to low, net revenue would only decrease slightly at first, and more steeply at low IHCs. This result is largely independent of the individual catch quota available.

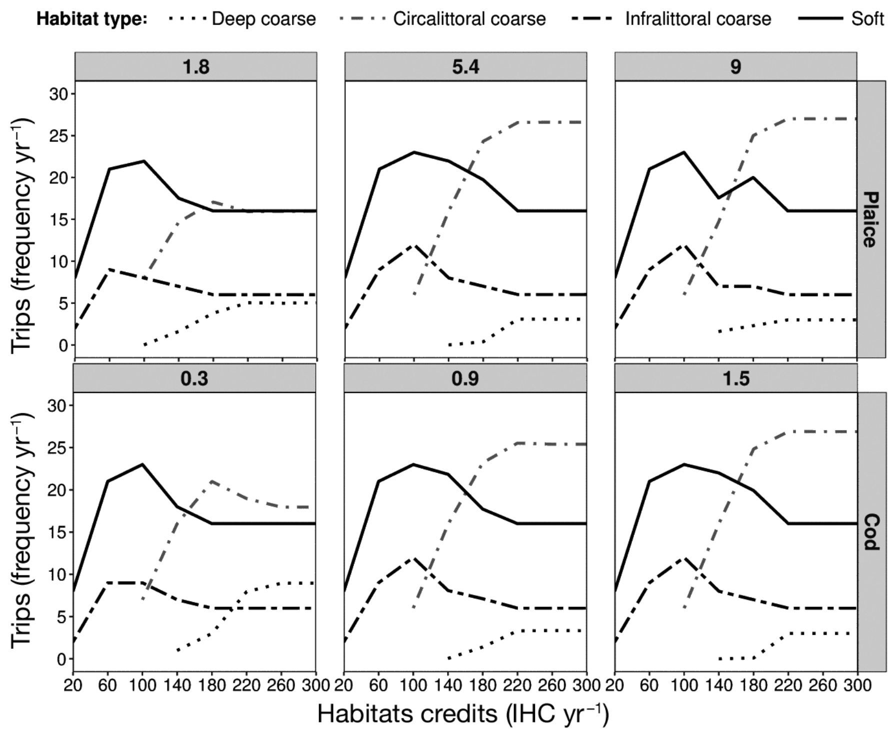

Fig. 7. Average number of trips made to a certain habitat type for the different impact scenarios. Top panels are for a fishery under plaice quota from 1.8 to 9 t yr^{-1}; lower panels are for the fishery constrained by cod quota from 0.3 to 1.5 t yr^{-1}

Trip allocation over habitat types

When habitat credits are combined with plaice quota, the choice of fishing grounds is determined by the IHC available and the level of the plaice quotas (Fig. 7). High IHC (>160) and a high plaice quota (>5 t) result in an allocation of trips predominantly to circalittoral coarse and soft sediment habitats, although a few vessels will fish in areas with more sensitive deep coarse sediments. A reduction in IHC (<160) pushes vessels away from circalittoral coarse sediments to less sensitive infralittoral coarse and soft sediments. The shift from more sensitive to less sensitive habitat types is influenced by the amount of individual plaice quota. Fishing in deep coarse sediments stops when habitat credits are reduced below 140 IHC. At a low plaice quota (<5 t), trips to deep coarse sediments decline but continue until IHC < 100 and trips to these sensitive habitats are no longer operated.

In addition to the IHC available, low cod quota influences the choice of fishing grounds (Fig. 7). At high IHC and low cod quota, vessels will allocate more trips to sensitive deep coarse sediment habitats and less trips to fishing grounds with circalittoral coarse sediment habitat. Nevertheless, circalittoral coarse and soft sediment habitats still remain the dominant fishing grounds. At high cod quota, fishing in more sensitive habitat types rapidly decreases if IHCs are reduced (Fig. 7). At low cod quota, however, a decrease in IHC initially results in a small increase in the number of trips to circalittoral fishing grounds and a decrease in the number of trips allocated to fishing grounds with deep coarse sediment habitats. A further decrease in IHC will push vessels away from sensitive habitat types, and as a result, all effort is concentrated in infralittoral coarse and soft sediment habitats.

DISCUSSION

Overview of findings

The model results indicate that managing fisheries by catch quota would not necessarily reduce the benthic impact exerted by French bottom trawlers in the Eastern English Channel. Reduction of individual catch quotas could even result in an increase in the benthic impact, as vessels could reallocate effort towards fishing grounds located in more vulnerable habitats to avoid catching quota-restricted species. In the model, we showed that benthic impacts can be reduced by introducing IHCs, with limited cost in terms of landings and revenue. Vessels are still able to reallocate their effort to less vulnerable fishing grounds while maintaining their effort and revenue. When IHCs are reduced further, vessels cannot make any profit out of fishing and they have to stay in port to not exceed their habitat credits.

Simulation model

The simulation model was parameterised for the Eastern English Channel French bottom trawl fleet. Results of the fleet response are indicative for this particular fleet given the necessary simplifications made. Model simplifications included the assumptions that (1) fishers are profit maximizers and comply fully with management regulations given severe sanctions for non-compliance, (2) individuals have perfect knowledge about the temporal and spatial distribution of catch rates, (3) the quota for a single species (plaice or cod) is constraining the fishery, and these quotas are not transferable, and (4) there is no competition among fishers. The hypothesized effects of these simplifying assumptions have been discussed in Poos et al. (2010) and Batsleer et al. (2013). We further assumed that the scallop season was open from November to April throughout the Eastern English Channel and that vessels did not move from Boulogne-Sur-Mer to other harbours at the opening of the scallop season. In reality, the opening of the season depends on annual scientific advice and the opinion of authorities, and may be different among areas (Carpentier et al. 2009). Also, vessels may temporarily move to harbours close to the best scallop fishing grounds.

Although we had to make a large number of simplifying assumptions, as discussed above, these assumptions will not affect the qualitative results, showing that within mixed fisheries the seasonally and spatially variable availability of target species allows a variety of effort allocation patterns that yield similar net revenue with widely different benthic impacts.

In this study, habitat impact credits were set independent of the historic fishing activities in the habitat. However, historic fishing activities can be an important factor determining fishing impact on the benthic ecosystem (Kaiser 2005). Reducing habitat credits in frequently fished areas may lead to a concentration of fishing activities in those areas, given the reduced cost of fishing (Holland & Schnier 2006). This reduces the risk of trawling in areas that were previously untrawled.

The habitat credits in the model do not vary by gear type. Empirical studies, however, indicate that gear type strongly affects the magnitude of fishing impact (Collie et al. 2000, Kaiser et al. 2006, Eigaard et al. 2016). Sensitivity matrices can synthesize these results, describing the sensitivity of habitats to different gears based on the physical bottom of each gear and the sensitivity of the benthic community to the additional mortality caused by the bottom contact. These sensitivity matrices can then be used by policy makers to assign gear-specific habitat credits (Eno et al. 2013).

Practical application and broader relevance

There is a growing awareness of the effects of fisheries on ecosystems, beyond the obvious reductions in population biomass as a result of the fisheries catches. An ecosystem-based approach to fisheries management aims at mitigating these effects. One aspect of ecosystem-based management is the recovery and conservation of the composition, structure and functioning of ecosystems. Management approaches, such as credit management systems, have been suggested as a means to protect or improve ecosystem structure and function (Holland & Schnier 2006, Kraak et al. 2012), as a complement to traditional measures such as closed areas (Winter & May 2001, Raakjær Nielsen 2003, Smith et al. 2010, Abbott & Haynie 2012, Caveen et al. 2015). Such systems build on incentivizing responsible fishing practices, constraining fishers with a number of credits or a share of the property while allowing them to adjust their behaviour and choose freely when and where to fish to make optimal use of their credits or share. However, management objectives in such systems will only be achieved if credits or quotas are set at the right target or limit reference point (Van Riel et al. 2015). Setting these targets or limit reference points will be a process in which policy-makers, scientists, the fishing industry, and other stakeholders need to be engaged and to agree on the goals of the management system. Successful implementation further depends on the availability of data and knowledge on habitats and their functioning, as well as detailed monitoring of vessel locations, and bookkeeping systems of credits in the fishing industry and government institutions.

Our study shows that a complementary use of habitat credits with a catch quota could reduce benthic impact, with the fishery continuing to sustainably exploit a mix of resources. The development of credit systems, however, is still in its infancy and requires more knowledge on the relationship between different ecosystem components, the relationship between human use components, improved data collection at the right spatial and temporal scale, and extensive monitoring and enforcement (Kraak et al. 2012, Tidd et al. 2015, Van Riel et al. 2015, Sys et al. 2016). The collection of more detailed information and data gathering would require more effort from the fishing industry in terms of privacy and investments on board, but fishers would still retain their freedom to choose when and where to fish. Hence, for spatial measures to meet ecological objectives and to be cost-effective, an understanding of the adaptive behaviour of fishers to these measures is critical.

Acknowledgements. The contribution of J.J.P. and A.D.R. was funded by the European Union's Seventh Framework Programme (FP7/2007-2013) under grant agreement no. 312088 (BENTHIS). The contribution of P.M., S.V., and Y.V. was funded by the European Union's Seventh Framework Programme (FP7/2007-2013) under grant agreement no. 266445 for the project Vectors of Change in Oceans and Seas Marine Life, Impact on Economic Sectors (VECTORS). We thank the 3 anonymous reviewers whose comments helped improve and clarify the manuscript.

LITERATURE CITED

Abbott JK, Haynie AC (2012) What are we protecting? Fisher behavior and the unintended consequences of spatial closures as a fishery management tool. Ecol Appl 22:762–777
Atkinson CJL, Bergmann M, Kaiser MJ (2004) Habitat selection in whiting. J Fish Biol 64:788–793
Batsleer J, Poos JJ, Marchal P, Vermard Y, Rijnsdorp AD (2013) Mixed fisheries management: protecting the weakest link. Mar Ecol Prog Ser 479:177–190
Boulcott P, Millar CP, Fryer RJ (2014) Impact of scallop dredging on benthic epifauna in a mixed-substrate habitat. ICES J Mar Sci 71:834–844
Carpentier A, Martin CS, Vaz S (eds) (2009) Channel habitat atlas for marine resource management, final report (CHARM phase II). INTERREG 3a Programme, IFREMER, Boulogne-sur-mer
Caveen A, Polunin N, Gray T, Stead SM (2015) The controversy over marine protected areas: science meets policy. Springer, London
Chu C (2009) Thirty years later: the global growth of ITQs and their influence on stock status in marine fisheries. Fish Fish 10:217–230
Clark CW, Mangel M (2000) Dynamic state variable models in ecology: methods and applications. Oxford University Press, Oxford
Coggan R, Diesing M (2011) The seabed habitats of the central English Channel: A generation on from Holme and Cabioch, how do their interpretations match-up to modern mapping techniques? Cont Shelf Res 31:S132–S150
Collie JS, Escanero GA, Valentine PC (1997) Effects of bottom fishing on the benthic megafauna of Georges Bank. Mar Ecol Prog Ser 155:159–172
Collie JS, Hall SJ, Kaiser MJ, Poiner IR (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. J Anim Ecol 69:785–798
Darling ES (2014) Assessing the effect of marine reserves on household food security in Kenyan coral reef fishing communities. PLOS ONE 9:e113614
Daurès F, Trenkel VM, Guyader O (2013) Modelling the fishing costs of French commercial vessels in the Bay of Biscay. Fish Res 146:74–85
Dayton PK, Thrush SF, Agardy MT, Hofman RJ (1995) Environmental effects of marine fishing. Aquat Conserv 5: 205–232
Delavenne J (2012) Conservation of marine habitats under multiple human uses: methods, objectives and constraints to optimize a marine protected areas network in the Eastern English Channel. PhD dissertation, Université du Littoral Côte d'Opale, Dunkerque
Diesing M, Coggan R, Vanstaen K (2009) Widespread rocky reef occurrence in the central English Channel and the implications for predictive habitat mapping. Estuar Coast Shelf Sci 83:647–658
Dinmore TA, Duplisea DE, Rackham BD, Maxwell DL, Jennings S (2003) Impact of a large-scale area closure on patterns of fishing disturbance and the consequences for benthic communities. ICES J Mar Sci 60:371–380
Dowling NA, Wilcox C, Mangel M, Pascoe S (2012) Assessing opportunity and relocation costs of marine protected areas using a behavioural model of longline fleet dynamics. Fish Fish 13:139–157
Eigaard OR, Bastardie F, Breen M, Dinesen GE and others (2016) Estimating seafloor pressure from trawls and dredges based on gear design and dimensions. ICES J Mar Sci 73(Suppl 1):i27–i43
Eno NC, Frid CLJ, Hall K, Ramsay K and others (2013) Assessing the sensitivity of habitats to fishing: from seabed maps to sensitivity maps. J Fish Biol 83:826–846
Gerritsen H, Lordan C (2011) Integrating vessel monitoring systems (VMS) data with daily catch data from logbooks to explore the spatial distribution of catch and effort at high resolution. ICES J Mar Sci 68:245–252
Gibson RN, Robb L (2000) Sediment selection in juvenile plaice and its behavioural basis. J Fish Biol 56:1258–1275
Gillis DM, Pikitch EK, Peterman RM (1995) Dynamic discarding decisions: foraging theory for high-grading in a trawl fishery. Behav Ecol 6:146–154
Guyader O, Daures F, Fifas S (2004) A bioeconomic analysis of the impact of decommissioning programs: application to a limited-entry French scallop fishery. Mar Resour Econ 19:225–242
Hall SJ (1994) Physical disturbance and marine benthic communities: life in unconsolidated sediments. Oceanogr Mar Biol Annu Rev 32:179–239
Halpern BS (2003) The impact of marine reserves: Do reserves work and does reserve size matter? Ecol Appl 13:117–137
Hiddink JG, Jennings S, Kaiser M (2007) Assessing and predicting the relative ecological impacts of disturbance on habitats with different sensitivities. J Appl Ecol 44: 405–413
Hintzen NT, Bastardie F, Beare D, Piet GJ and others (2012) VMStools: open-source software for the processing, analysis and visualisation of fisheries logbook and VMS data. Fish Res 115-116:31–43
Holland D, Schnier KE (2006) Individual habitat quotas for fisheries. J Environ Econ Manage 51:72–92
Houston AI, McNamara JM (1999) Models of adaptive behaviour. Cambridge University Press, Cambridge

Jennings S, Kaiser MJ (1998) The effects of fishing on marine ecosystems. Adv Mar Biol 34:201–352
Kaiser MJ (2005) Are marine protected areas a red herring or fisheries panacea? Can J Fish Aquat Sci 62:1194–1199
Kaiser MJ, Collie JS, Hall SJ, Jennings S, Poiner IR (2002) Modification of marine habitats by trawling activities: prognosis and solutions. Fish Fish 3:114–136
Kaiser MJ, Clarke KR, Hinz H, Austen MCV, Somerfield PJ, Karakassis I (2006) Global analysis of response and recovery of benthic biota to fishing. Mar Ecol Prog Ser 311:1–14
Kaiser MJ, Hilborn R, Jennings S, Amaroso R and others (2016) Prioritization of knowledge-needs to achieve best practices for bottom trawling in relation to seabed habitats. Fish Fish 17:637–663
Kraak SBM, Reid DG, Gerritsen HD, Kelly CJ, Fitzpatrick M, Codling EA, Rogan E (2012) 21st century fisheries management: a spatio-temporally explicit tariff-based approach combining multiple drivers and incentivising responsible fishing. ICES J Mar Sci 69:590–601
Kraak SBM, Bailey N, Cardinale M, Darby C and others (2013) Lessons for fisheries management from the EU cod recovery plan. Mar Policy 37:200–213
Le Pape O, Chauvet F, Mahévas S, Lazure P, Guérault D, Désaunay Y (2003) Quantitative description of habitat suitability for the juvenile common sole (*Solea solea*, L.) in the Bay of Biscay (France) and the contribution of different habitats to the adult population. J Sea Res 50: 139–149
Lester SE, Halpern BS, Grorud-Colvert K, Lubchenco J and others (2009) Biological effects within no-take marine reserves: a global synthesis. Mar Ecol Prog Ser 384: 33–46
Lindholm JB, Gleason M, Kline D, Clary L, Rienecke S, Cramer A, Los Huertos M (2015) Ecological effects of bottom trawling on the structural attributes of fish habitat in unconsolidated sediments along the central California outer continental shelf. Fish Bull 113:82–96
Pikitch EK, Santora C, Babcock EA, Bakun A and others (2004) Ecosystem-based fishery management. Science 305:346–347
Poos JJ, Bogaards JA, Quirijns FJ, Gillis DM, Rijnsdorp AD (2010) Individual quotas, fishing effort allocation, and over-quota discarding in mixed fisheries. ICES J Mar Sci 67:323–333
R Core Team (2013) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna
Raakjær Nielsen J (2003) An analytical framework for studying: compliance and legitimacy in fisheries management. Mar Policy 27:425–432
Rice J, Arvanitidis C, Borja A, Frid C and others (2012) Indicators for sea-floor integrity under the European Marine Strategy Framework Directive. Ecol Indic 12:174–184
Rijnsdorp AD, Daan N, Dekker W (2006) Partial fishing mortality per fishing trip: a useful indicator of effective fishing effort in mixed demersal fisheries. ICES J Mar Sci 63: 556–566
Rijnsdorp AD, Poos JJ, Quirijns FJ (2011) Spatial dimension and exploitation dynamics of local fishing grounds by fishers targeting several flatfish species. Can J Fish Aquat Sci 68:1064–1076
Rijnsdorp AD, Bastardie F, Bolam SG, Buhl-Mortensen L and others (2015) Towards a framework for the quantitative assessment of trawling impacts on the sea bed and benthic ecosystem. ICES J Mar Sci 73(Supp 1):i127–i138 doi:10.1093/icesjms/fsv207
Smith MD, Lynham J, Sanchirico JN, Wilson JA (2010) Political economy of marine reserves: understanding the role of opportunity costs. Proc Natl Acad Sci USA 107: 18300–18305
Snickars M, Gullström M, Sundblad G, Bergström U, Downie AL, Lindegarth M, Mattila J (2014) Species-environment relationships and potential for distribution modelling in coastal waters. J Sea Res 85:116–125
Sys K, Poos JJ, Van Meensel J, Polet H, Buysse J (2016) Competitive interactions between two fishing fleets in the North Sea. ICES J Mar Sci 73:1485–1493
Thouzeau G, Robert G, Smith SJ (1991) Spatial variability in distribution and growth of juvenile and adult sea scallops *Placopecten magellanicus* (Gmelin) on eastern Georges Bank (Northwest Atlantic). Mar Ecol Prog Ser 74: 205–218
Thrush SF, Hewitt JE, Cummings VJ, Dayton PK and others (1998) Disturbance of the marine benthic habitat by commercial fishing: impacts at the scale of the fishery. Ecol Appl 8:866–879
Tidd AN, Vermard Y, Marchal P, Pinnegar J, Blanchard JL, Milner-Gulland EJ (2015) Fishing for space: fine-scale multi-sector maritime activities influence fisher location choice. PLOS ONE 10:e0116335
Tupper M, Boutilier RG (1995) Effects of habitat on settlement, growth, and post-settlement survival of Atlantic cod (*Gadus morhua*). Can J Fish Aquat Sci 52:1834–1841
Valdemarsen JW, Jørgensen T, Engås A (2007) Options to mitigate bottom habitat impact of dragged gears. FAO Fish Tech Pap 506:1–29
van Denderen PD, Hintzen NT, Rijnsdorp AD, Ruardij P, van Kooten T (2014) Habitat-specific effects of fishing disturbance on benthic species richness in marine soft sediments. Ecosystems 17:1216–1226
van Denderen PD, Bolam SG, Hiddink JG, Jennings S, Kenny A, Rijnsdorp AD, van Kooten T (2015) Similar effects of bottom trawling and natural disturbance on composition and function of benthic communities across habitats. Mar Ecol Prog Ser 541:31–43
Van Riel MC, Bush SR, van Zwieten PAM, Mol APJ (2015) Understanding fisheries credit systems: potentials and pitfalls of managing catch efficiency. Fish Fish 16: 453–470
Wieland K, Pedersen EMF, Olesen HJ, Beyer JE (2009) Effect of bottom type on catch rates of North Sea cod (*Gadus morhua*) in surveys with commercial fishing vessels. Fish Res 96:244–251
Winter SC, May PJ (2001) Motivation for compliance with environmental regulations. J Policy Anal Manage 20: 675–698
Wood SN (2003) Thin plate regression splines. J R Stat Soc Series B Stat Methodol 65:95–114
Wood SN (2006) Generalized additive models: an introduction with R. CRC Press, Boca Raton, FL
Zuur AF (2012) A beginners guide to generalized additive models with R. Highland Statistics, Newburgh
Zuur AF, Ieno EN, Walker NJ, Saveliev AA, Smith GM (2009) Mixed effect models and extensions in ecology in R. Springer, New York, NY

Editorial responsibility: Jake Rice, Ottawa, Ontario, Canada

Submitted: April 21, 2017; Accepted: October 30, 2017
Proofs received from author(s): December 19, 2017