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# An integrated methodology for assessing ecological and economic impacts for marine management: A case study for abrasion and mobile fishing gear effects

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ABSTRACT

Ensuring a balance of conservation and human activities within the marine environment requires complex decisions on how to achieve conservation objectives whilst considering the likely economic and social costs and benefits of such decisions. We assessed the risk of adverse effects from abrasion caused by bottom fishing, as a case study, to demonstrate an integrated spatial management methodology. Risk was quantified in the Galway Bay, Ireland region using a 1 km grid, as the ratio between mortality and recovery of benthic infauna exposed to abrasion effects estimated from fishing data. Risk was considered high if annual mortality exceeded recovery. A Bayesian network model, incorporating ecological and economic data, compared three management scenarios: i) business as usual (BAU), ii) exclusion of fishing vessels over 18 m within 6 nautical miles (68% of study area) (Length) and iii) a network of marine protected areas (MPAs) closed to all fishing vessels (11% of study area) (MPA). The BAU scenario indicated that risk was high for 85.6% of the study area. The length scenario resulted in a 14.6% reduction in high risk cells, affecting €0.9 M of landings, €0.7 M of indirect output and 19 full time equivalent (FTE) employees. The MPA scenario resulted in a 1.7% reduction of high risk cells. This scenario affected €0.17M in landings, €0.13M in indirect outputs and 3 FTE employees. We present a method which provides a quantitative, spatial, risk and economic impact assessment of the effects of human activities and pressures on the marine environment that assesses potential costs and benefits of management options to mitigate against these pressures. Future development of a mortality-recovery ratio for defined time periods would enable application across multiple habitats, human activities and pressures.

## 1. Introduction

Over the last twenty years, marine management has been moving toward a more ecosystem-based approach which recognises all components and threats to the ecosystem rather than a single issue/activity or species (Garcia et al., 2003; Katsanevakis et al., 2011; Borgwardt et al., 2019; Lago et al., 2019). One of the fundamental aims behind ecosystem-based management (EBM) is that it achieves excellent ecological status as well as ensuring sustainable and economically viable outputs and maintains the provision of important goods and services from the marine environment (Katsanevakis et al., 2011). EBM principals form the basis of several European legislations, such as the revised Common Fisheries Policy (EC, 2013), the Marine Strategy Framework Directive (MSFD) (EC, 2008) and the Maritime Spatial Planning (MSP) Directive (EC, 2014). These directives require the consideration of all ecosystem components when managing marine activities, making ecosystem assessments complex.

The large number of marine activities and the pressures which they exert upon ecosystem components have been collectively mapped (Halpern et al., 2008) and assessed at broad spatial scales using qualitative and semi-qualitative methodologies (Breen et al., 2012; Knights et al., 2013, 2015; Goodsir et al., 2015; Piet et al., 2015, 2019; Stelzenmüller et al., 2018; Borgwardt et al., 2019). These methods employ a risk-based approach, with scoring based on information on the likely impact of the pressure and the ecosystem capacity to recover from the impact (Knights et al., 2015). Similar risk-based approaches have been employed in single activity-pressure descriptors across a broad range of ecosystem components (Hobday et al., 2011; Brown et al., 2013, 2015; Breen et al., 2017). These types of assessments are useful within a hierarchical framework for cost-effectively highlighting the activities and pressures and their spatial and temporal distributions within an ecosystem (Smith et al., 2007). However, the lack of quantitative data on

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risk and the likely comparative effects of a range of potential management measures and solutions limit the implementation potential when it comes to decision making for regulatory authorities. Thus ensuring a balance of conservation and human activities across all facets of the marine environment requires regulatory authorities to make decisions on how to achieve conservation objectives whilst considering the likely economic and social costs and benefits of such decisions.

Management Strategy Evaluation (MSE) is one tool which has been used to assess the trade-offs between the ecological effects and economic costs/benefits of alternative management options. MSE, for EBM assessment, uses ecosystem models to run simulations to assess performance measures or indicators against a range of different management strategies (Carruthers et al., 2014; Fulton et al., 2014; Hordyk et al., 2015). Whilst this simulation data driven approach was shown in Fulton et al. (2014) to be a useful tool to assess multiple management strategies, its current use is limited to commercially exploited fisheries management where information on age, size, population structure are used to assess different fisheries management options. There is a need to develop empirical approaches which allow for a broader range of human activities and pressures to be assessed and to incorporate both ecological and economic factors into management option evaluation.

A Bayesian network model is a probabilistic reasoning tool which learns probabilistic relations from data which can then be used to make decisions and predictions on future scenarios. Such models have been combined with a GIS framework to support marine spatial planning (Stelzenmüller et al., 2010; Coccoli et al., 2018; Pınarbaşı et al., 2019) and to compare fisheries management options (Stelzenmüller et al., 2015). Probabilistic reasoning is a useful and simple tool to assess risks and is better than models which only account for expected values (Uusitalo, 2007). Probabilistic reasoning's ability to predict both causes and consequences, incorporate expert judgement where empirical data is lacking, combine data from different sources, deal with both ecological and economic data, and their efficient computation time make them a valuable tool within marine fisheries management (Uusitalo, 2007).

One of the key activities which consistently ranks as high priority for management in ecosystem assessments is abrasion on the seafloor caused by bottom fishing activities (Goodsir et al., 2015; Knights et al., 2015). Under the MSFD descriptor 6 on seafloor integrity, indicator criterion have been developed which pertains to the 'spatial extent and distribution of physical disturbance of pressures on the seabed' (EC, 2008). This criterion requires member states to implement management measures to ensure that the spatial extent of activities physically damaging to the benthos do not exceed what would be expected under 'good environmental status'. One of the ways to achieve this would be to implement spatial restrictions on benthic fishing activities within certain areas.

Considering the requirement for EBM, the need for quantitative methods to assess risk from marine pressures, and decision support tools to enable and optimise marine spatial planning, we build on the existing literature on Bayesian network analysis and developed a spatially explicit, quantitative and multidisciplinary assessment of options for spatial management of human pressures in the marine environment. The scenarios presented reflect the types of decisions that marine regulatory authorities and marine spatial planners might face and demonstrate a way forward for comprehensively assessing the effects of multiple marine management options.

## 2. Methods

### 2.1. Study area

Galway Bay area is located on the west coast of Ireland (Fig. 1). The study area extends from the high water mark seaward to the 12 nm limit with a north and south border at two headlands either side of the greater bay area (Fig. 1). The area contains economically important *Nephrops* within a large area of muddy sediment and demersal fishing grounds and the presence of sensitive habitats such as seagrass and mearl beds in the inshore bays have led to these areas being designated Special Areas of Conservation (SACs) under the European Union's Habitat Directive (EC, 1992) (Fig. 1).

Table 1 shows code and habitat descriptions. The blue lines represent existing Special Areas of Conservation in the study area. Inset map shows study area location in the west coast of Ireland.

We established a 1 km × 1 km grid over the study area using the fishnet tool in ArcGIS 10.3. The 1 km resolution was chosen as the best size which would represent habitat heterogeneity in the study area

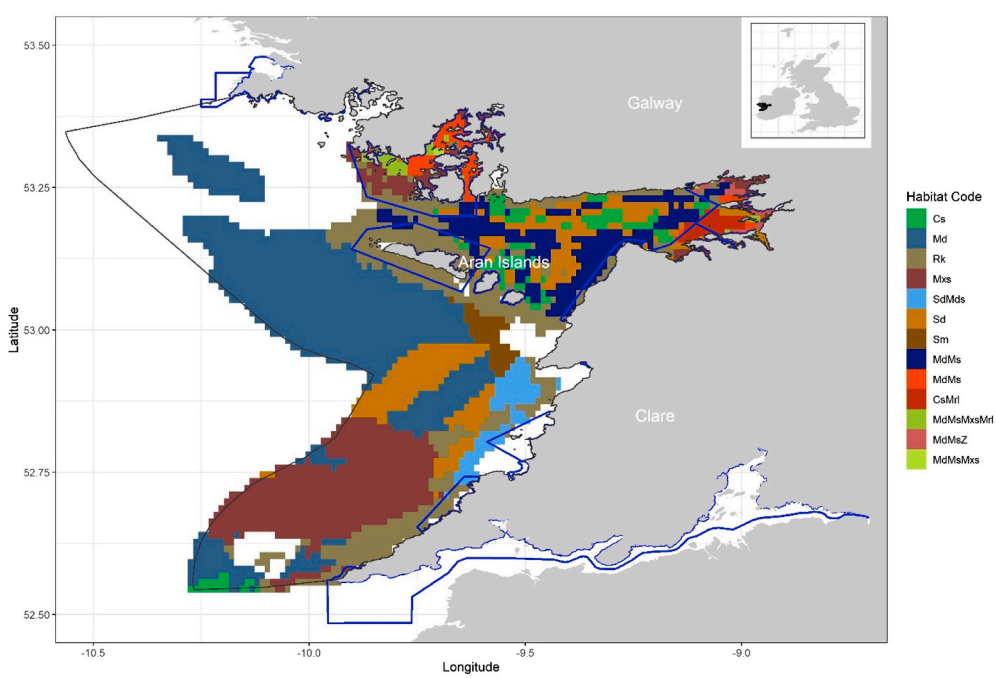


**Fig. 1.** Map showing the study area (black boundary) on the west of Ireland. The study area extends out to 12 nm (outer black boundary) with a north and south border at two headlands either side of the bay. The combined habitat classifications are overlayed within the study area.

whilst keeping as close as possible to available Vessel Monitoring System (VMS) resolution (see below).

### 2.2. Data

#### 2.2.1. Habitat data

Modelled habitat data was available from the EMODnet project as modelled broad scale habitat layers (http://www.emodnet.eu/seabed-habitats). Habitat maps within SACs, derived from interpolation of point and transect based surveys, were available from the National Parks and Wildlife Service (Marine Institute, 2015). The Galway Bay SAC habitat data was considered the most accurate dataset as this was generated from survey data. These data were therefore used when available and accounted for about 58% of the total mapped area. For the rest of the area, habitat data was filled in using the modelled EMODnet dataset. Habitat types from the different sources were combined based on standard EUNIS Level 3 codes (Table 1). Each grid cell was assigned a habitat on a 1 km grid resolution based on two rules: 1. if maerl or seagrasses were present in the grid cell then that habitat was assigned to that grid cell and 2. the habitat with the highest proportion within the cell took priority. This ensured that sensitive habitats which may exist in small patches were not lost in the subsequent analysis.

#### 2.2.2. Vessels and vessel monitoring system (VMS) data

We estimated the average annual (2014–2016) spatial distribution of fishing effort from vessel monitoring system (VMS) and landing declarations (logbooks) data for four Irish demersal fishing vessels >12 m in length (Fig. 2). VMS reporting is mandatory for vessels >12 m, is fully implemented in Irish waters since 2012 and provides the position, course and speed of the vessel with a typical time resolution of 2 h. Vessels under 12 m are not required to have VMS on-board therefore effort information for this fleet was not available. The four fleets assessed were, otter trawls 12–15 m in length (OTB 12–15 m), otter trawls 15–18 m in length (OTB 15–18 m), otter trawls over 18 m (OTB >18 m) and seine nets over 18 m (SSC >18 m). These fleets account for all bottom towed fishing gears in the area. There is no dredging or beam trawling by vessels over 12 m in the area.

VMS and logbook data can be linked, using particular allocation rules, to provide more precise location information for fishing effort and landings (Bastardie et al., 2010; Gerritsen and Lordan, 2010; Hintzen et al., 2012). These linked logbook and VMS data have been used previously to estimate the fishing locations of the Irish bottom otter trawling fleet (Hynes et al., 2016). The most likely fishing gear was attributed to the VMS pings for a fishing trip based on the logbook data. Then, following filtering of pings based on gear specific fishing speeds (Gerritsen and Lordan, 2010) pings were classed as "fishing pings" or non-fishing pings and the landings were allocated among fishing pings of the same day and trip (in that order of priority), following the methods in Hintzen et al. (2012).

To estimate the fishing footprint (swept area), the vessel track length associated to each ping was estimated as the mean speed, calculated based on the distance and time difference of the two adjacent pings (forward-backward mean speed), multiplied by the ping interval. The surface and subsurface seabed contact width for towed gears were estimated using the gear inferred from the logbook and vessel length (LOA, m) and power (KW) from the EU fleet register, following the relationships estimated by Eigaard et al. (2015, 2016).

For Scottish Seines (SSC), the footprint estimated by Eigaard et al. (2015, 2016) was relative to a single fishing event. However, VMS data with a time resolution of 2 h do not allow for the identification of individual fishing events, only the detection of fishing activity. Assuming the fishing events to be placed end to end, the mean surface contact width was estimated as L/6, where L is the estimated length of the seine ropes (based on a corrected estimation of the contact surface in their supplementary material):

$$Surface - contact\ area = \frac{1}{2} * \left(\frac{L}{3}\right) * 2H \quad (1)$$

where $H$ is the height of the triangle made by the track of the hauled

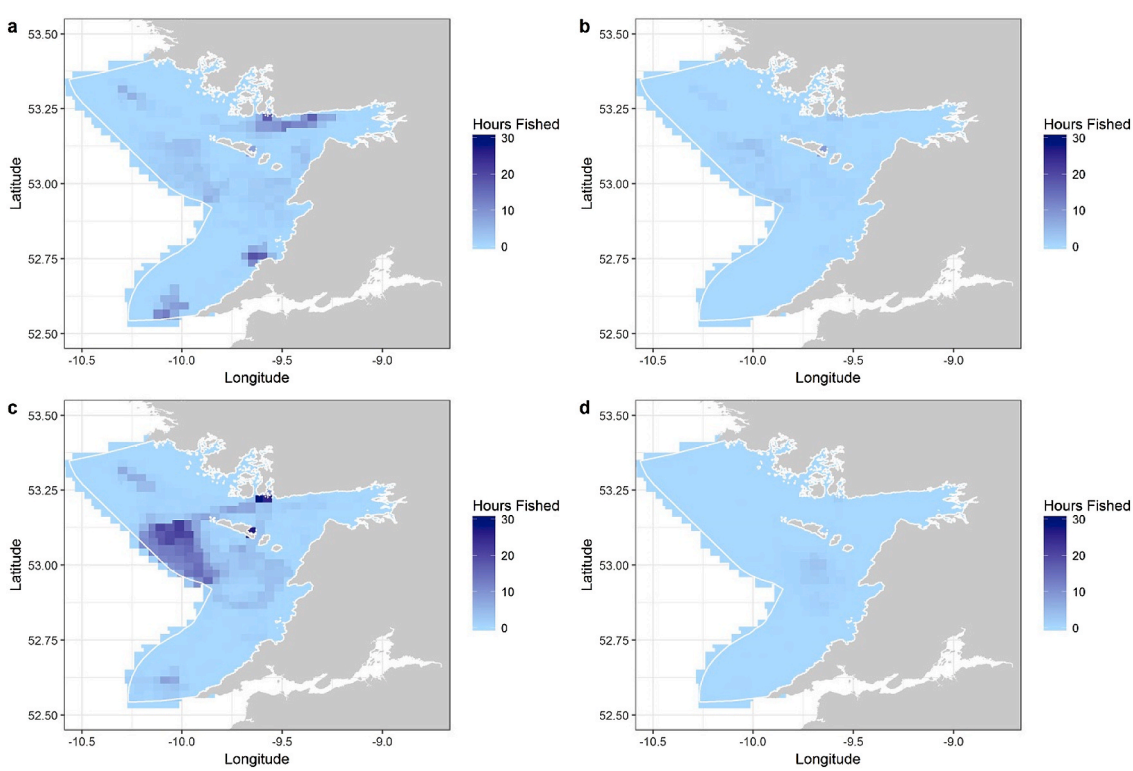


**Fig. 2.** Hours fished per grid cell per annum averaged across 2014–2016 from Vessel Monitoring System data; a) Otter trawls between 12 and 15 m, b) Otter trawls between 15 and 18 m, c) Otter trawls over 18 m and d) Seine nets over 18 m.

**Table 1**
The combined habitat codes used for the study including a description of the EUNIS classes which contributed to each habitat class. Proportional information on mud, sand, gravel, muddy sand, maerl, seagrass and rock for each habitat type are included as well as recovery and mortality values per habitat calculated for the risk assessment. $RT_{Hab}$ = Total recovery time for the habitat, $Rfr_{Hab}$ = Total recovery frequency for the habitat, $R_{Hab}$ = Total recovery per habitat, $MR_{Hab}$ = Total mortality for the habitat.

| Name | Coarse Sediments (Cs) | Coarse sediments with maerl (CsMrl) | Mixed sediments (Mxs) | Mud (Md) | Mud and muddy sand (MdMs) | Mud, muddy sand and mixed sediments (MdMsMxs) | Mud, muddy sand, mixed sediments and maerl (MdMsMxsMrl) | Mud, muddy sand and Zostera (MdMsZ) | Rock (Rk) | Sand (Sd) | Sand and muddy sand (SdMds) | Sandy mud (Sm) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| EUNIS codes | A5.13 A5.14 A5.15 | A5.13 A5.14 A5.15 | A5.15 A5.43 A5.44 | A5.36 A5.37 | A5.23 A5.24 A5.25 A5.26 | A5.23 A5.24 A5.25 A5.26 | A5.23 A5.24 A5.25 A5.26 | A5.23 A5.24 A5.25 A5.26 | A3.1 A3.3 A4.1 A4.2 A4.27 | A5.33 A5.35 A5.27 | A5.35 | A5.27 |
| Prop. Mud | – | – | 0.25 | 1 | 0.2 | 0.2 | 0.2 | 0.2 | – | 0.2 | 0.2 | – |
| Prop. Sand | 0.5 | 0.33 | 0.25 | – | 0.4 | 0.267 | 0.2 | 0.2 | – | 0.4 | – | 0.5 |
| Prop. Gravel | 0.5 | 0.33 | 0.25 | – | – | 0.267 | 0.2 | 0.2 | – | – | – | – |
| Prop. Muddy sand | – | – | 0.25 | – | 0.4 | 0.267 | 0.2 | 0.2 | – | 0.4 | 0.8 | 0.5 |
| Prop. Maerl | – | 0.33 | – | – | – | – | 0.2 | – | – | – | – | – |
| Prop. Seagrass | – | – | – | – | – | – | – | 0.2 | – | – | – | – |
| Prop. Rock | – | – | – | – | – | – | – | – | 1.00 | – | – | – |
| $RT_{Hab}$ (year) | 0.65 | 0.76 | 0.48 | 0.07 | 0.35 | 0.50 | 0.58 | 0.58 | 1.00 | 0.35 | 0.44 | 0.42 |
| $Rfr_{Hab}$ (year-1) | 2.17 | 1.76 | 5.13 | 14.29 | 4.95 | 4.52 | 4.30 | 4.30 | 1.00 | 4.95 | 4.37 | 2.61 |
| $R_{Hab}$ | 0.85 | 0.87 | 0.94 | 0.61 | 0.84 | 0.93 | 0.96 | 0.96 | 0.90 | 0.84 | 0.89 | 0.70 |
| $MR_{Hab}$ (proportion) | 0.64 | 0.65 | 0.57 | 0.35 | 0.55 | 0.59 | 0.60 | 0.53 | 0.29 | 0.55 | 0.61 | 0.61 |

**Table 2**
Per sediment values for recovery and decline taken from the literature. Recovery values were taken from Hiddink et al. (2006), Stelzenmuller et al. (2015), Freese et al. (1999), Hall-Spencer and Moore (2000), Neckles et al. (2005) and Foden et al. (2010). Literature for mortality values were Fock et al. (2011), Neckles et al. (2005), Hall-Spencer and Moore (2000) and Tillin et al. (2010). R = recovery time for sediment type, R*fr* = recovery frequency for sediment type, Decline = mortality per sediment type.

| Sediment parameter | Sediment value |
|---|---|
| $R_{Mud}$ (1year) | 0.07 |
| $R_{Sand}$ (1year) | 0.3 |
| $R_{Gravel}$ (1year) | 1 |
| $R_{MuddySand}$ (1year) | 0.53 |
| $R_{Maerl}$ (1year) | 1 |
| $R_{seagrass}$ (1year) | 1 |
| $R_{Rock}$ (1year) | 1 |
| $Rfr_{Mud}$ (year$^{-1}$) | 14.3 |
| $Rfr_{Sand}$ (year$^{-1}$) | 3.3 |
| $Rfr_{Gravel}$ (year$^{-1}$) | 1.0 |
| $Rfr_{MuddySand}$ (year$^{-1}$) | 1.9 |
| $Rfr_{Maerl}$ (year$^{-1}$) | 1.0 |
| $Rfr_{Seagrass}$ (year$^{-1}$) | 1.0 |
| $Rfr_{Rock}$ (year$^{-1}$) | 1.0 |
| $Decline_{Mud}$ *(proportion)* | 0.345 |
| $Decline_{Sand}$ *(proportion)* | 0.535 |
| $Decline_{Gravel}$ *(proportion)* | 0.74 |
| $Decline_{MuddySand}$ *(proportion)* | 0.675 |
| $Decline_{Maerl}$ *(proportion)* | 0.7 |
| $Decline_{Seagrass}$ *(proportion)* | 0.33 |
| $Decline_{Rock}$ *(proportion)* | 0.285 |

Recovery frequency per sediment ($Rfr_{Sediment}$) was calculated by dividing the recovery time into one year.

Seine. The swept area was then calculated, at each ping location, as the product of the estimated average surface or subsurface contact width and track length (ping interval times backward-forward mean speed).

The annual fishing effort in hours (h), landing value (€) and swept area (SA, km$^2$) were aggregated on a 3 km × 3 km grid as the sum of interval, value and swept area respectively across individual pings. The 3 km grid cells were subsequently resampled in ArcGIS 10.3 to 1 km grid cells and the cell values divided by 9 in order to align with the resolution used for habitat data (Fig. 2).

Swept area is a sum of individual fishing event footprints, not accounting for possible overlaps, and can therefore exceed the cell area. The swept-area ratio (SAR = SA/CA, with CA the cell area) over a given time range can be interpreted as a mean frequency of seabed impact by bottom gears within the cell (Gerritsen et al., 2013; Stelzenmüller et al., 2015), and is thereafter referred to as frequency swept (year$^{-1}$). No correction was applied to cell area where land overlapped, which means that the frequency swept may be underestimated for cells which intersect the coast, however, bottom trawling activity for over 12 m vessels in coastal cells in the study area is low.

### 2.3. Ecological risk assessment

We calculated the risk as the disturbance indicator for each grid cell ($DI_i$) following the methods developed by Fock (2011) and Stelzenmuller et al. (2015). It indicates the relative balance of (local mortality ($M_i$) and recovery ($R_i$) such that:

$$DI_i = M_i / R_i \quad (2)$$

$DI_i$ indicates the disturbance in cell *i* and is a unitless ratio where $DI_i = 1$ indicates that mortality and recovery are equal, $DI_i > 1$ indicates that mortality is greater than recovery and $DI_i < 1$ indicates that recovery is greater than mortality (Fock, 2011; Stelzenmüller et al., 2015). The advantage of a unitless ratio is that it allows for comparable measures between different types of pressures thereby accommodating the addition of other pressures into the assessment at a later stage. It also allows

more flexibility in situations where comparable rates for mortality and recovery are not available (Fock, 2011).

The combined relative recovery per habitat ($R_i$) was calculated using two parameters: the recovery time ($RT_{Hab}$) and the recovery frequency ($Rfr_{Hab}$). Recovery time is the portion of one year it takes the habitat to recover from one fishing event. Recovery frequency is the number of times in one year a habitat can recover from a single fishing event and is used to standardise recovery to the same (i.e., annual) temporal scale as the mortality calculation. The value for each habitat was combined with information taken from the literature about the portion of typical sediment types within each habitat (Table 1) and the recovery time for typical sediments ($RT_{Hab}$) (Table 2) (Stelzenmüller et al., 2015).

Equations for $RT_{Hab}$ and $Rfr_{Hab}$ were:

$$RT_{Hab} = \sum RT_{Sediment} * \text{proportion sediment} \quad (3)$$

$$Rfr_{Hab} = \sum Rfr_{Sediment} * \text{proportion sediment} \quad (4)$$

We calculated relative recovery per habitat ($R_i$) to 90% of pre-trawling condition for each grid cell *i*, using the equation developed in Fock (2011) and adapted by Stelzenmuller (2015):

$$R_i = 1 - (1 - 0.9RT_{Hab})^{Rfr_{Hab}} \quad (5)$$

Data for recovery times per year for mud, sand, muddy sand and gravel were taken from Hiddink et al. (2006). Gravel, seagrass, maerl and rock were reported in the literature as having recovery times greater than 1 year (Freese et al., 1999; Hall-Spencer and Moore, 2000; Neckles et al., 2005; Foden et al., 2010) therefore recovery for these sediment types was set at the maximum value of 1 in order to align with the annual temporal resolution for fishing effort data (Table 2).

Mortality per habitat ($M_{Hab}$) was defined as the percentage mortality per habitat after one trawling event, and similar to recovery was calculated using the average percentage decline of abundance per sediment type based on information from the literature. Values of average percentage decline of abundance per sediment type were taken from Stelzenmuller et al. (2015) and Fock et al. (2011) for mud, sand, muddy sand and gravel. Decline in seagrass abundance (0.33) was taken as an average of the four sites studied in Neckles et al. (2005). Maerl was given the value of 0.7 and taken from Hall-Spencer and Moore (2000) and Tillin et al. (2010) (Table 2).

Mortality per habitat was calculated similarly to recovery using information about the portion of sediments per habitat type and the mortality per sediment ($M_{Sediment}$):

$$M_{Hab} = \sum M_{Sediment} * \text{proportion sediment} \quad (6)$$

Annual fleet specific mortality per habitat type ($M_{ik}$) was calculated per fleet (*k*) and per cell (*i*) as:

$$M_{ik} = 1 - (1 - M_{Hab})F^{frik} \quad (7)$$

as outlined in Fock (2011) and Stelzenmuller et al. (2015), where $F_{frik}$ is the frequency swept per grid cell calculated as described in section 2.2.2. Annual local mortality for all fleets combined was calculated as the sum total of fleet specific mortality:

$$M_i = \sum M_{ik} \quad (8)$$

Finally, the disturbance indicator or risk per cell was calculated using equation (2) (Fock, 2011; Stelzenmüller et al., 2015).

### 2.4. Economic assessment

Through a multiplier effect, the output of other sectors in the economy may increase as a result of initial fishing activity (Grealis et al., 2017) therefore the landing value (LV), indirect output (IO), and full-time equivalent (FTE) employment were calculated across the entire study area. Landing value per cell was estimated as described in section 2.2.2 and summed across the study area using the zonal statistics tool in ArcGIS 10.3. Indirect output value is the increased business to business spending which results from the income generated from landings (Grealis and O'Donoghue, 2015). Indirect output was estimated using the output multiplier for commercial fisheries as modelled in the Bio-Economy Input-Output model (Grealis and O'Donoghue, 2015). Since the multiplier (*fisheries output multiplier = 1.77*) is the ratio between direct and indirect output, the multiplier was modified to reflect only the indirect output. The indirect output for the case study is:

$$IO = 0.77 * LV \quad (9)$$

Employment rates as FTE were also estimated using the employment multiplier for commercial fisheries from Grealis and O'Donoghue (2015). The employment multiplier (*employment multiplier = 1.5*) is the ratio between both indirect employees and direct employees that are supported from the change in landing value output. In order to calculate the FTE, the direct employees were estimated first using the jobs per million ratio (14 FTE/€*m*) for commercial fisheries derived in Grealis and O'Donoghue (2015a). The full-time equivalent of the case study was calculated as:

$$FTE = 1.5 * (14 * (LV/106)) \quad (10)$$

### 2.5. Bayesian network modelling

In order to integrate the risk assessment and economic analysis into a decision support framework that would allow for ecological and economic costs and benefits to be assessed together, therefore supporting a holistic approach to decision making for marine planning and management, we used a Bayesian network modelling software tool (Netica; www.norsys.com) to construct a Bayesian network of our ecological and economic assessments.

Variables were added to the model based on input data (i.e. habitats and fishing effort), risk assessment parameters (i.e. recovery and mortality) and output variables (i.e. landings value and risk). Data for each variable was discretised depending upon the distribution of values for

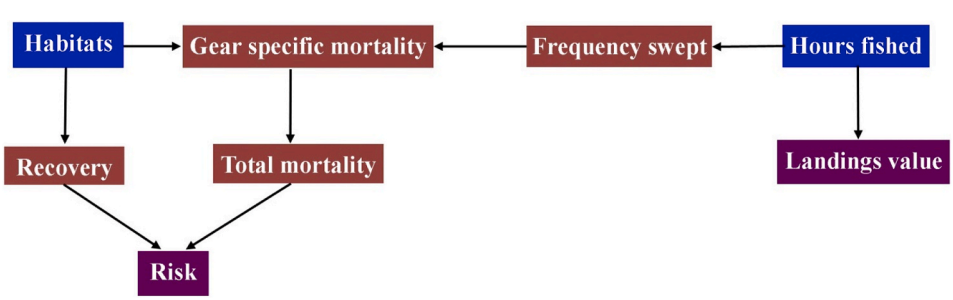


**Fig. 3.** Simplified structural specification of a Netica model showing the nodes and linkages used to build the model. Blue boxes denote input nodes, red boxes denote intermediate boxes used to generate risk and purple boxes denote output nodes. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

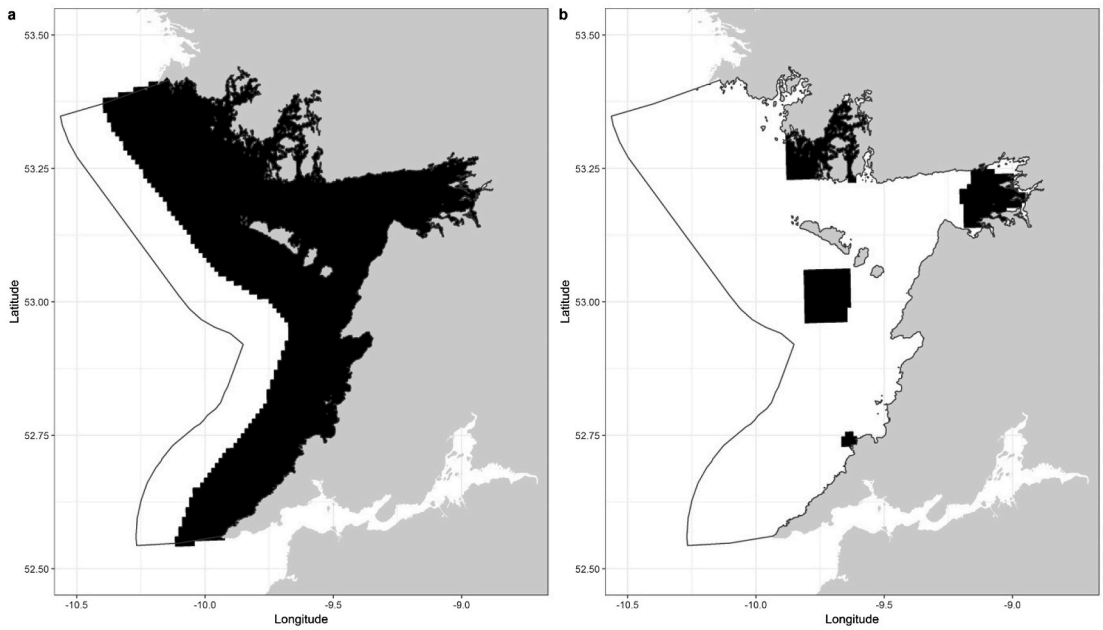


**Fig. 4.** The two management scenarios developed with black polygons denoting the area to be managed/closed. a) the area of exclusion for all vessels over 18 m in length and b) the 'best' Marxan solution showing the network of marine protected areas which protected 100% of the sensitive habitats and 10% of the representative habitats. Mobile fishing gears are completely excluded from these areas.

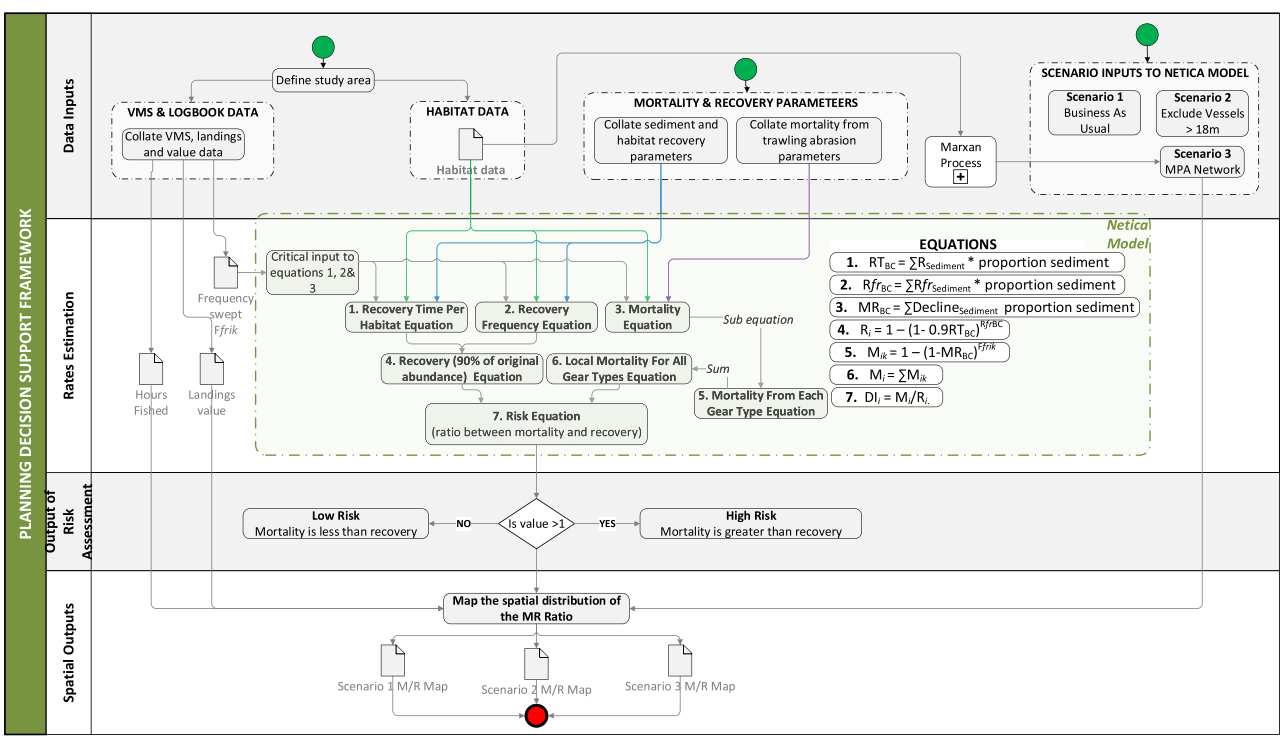


**Fig. 5.** Process flow diagram of methodological approach. Green circles show input data start points and red circle shows the end point. Equations are those used to calculate risk. The equation number corresponds to the task number showed in the flow diagram of the risk assessment. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

each dataset. Linkages were added depending upon the relationship between variables, for example, the relationship between habitats and recovery values. Conditional probability tables were populated based on these relationships using the data generated from the inputs and risk assessment. These tables reflect the probabilities of the cell having a value within each variable category given the configuration of parent values. A simplification of the final Netica model is shown in Fig. 3.

### 2.5.1. Scenarios

We developed three scenarios in order to test how different management options are likely to affect ecological risk and economic change.

*2.5.1.1. Business as usual (BAU scenario).* This reflects the current amount of trawling within the study area and used original fishing data

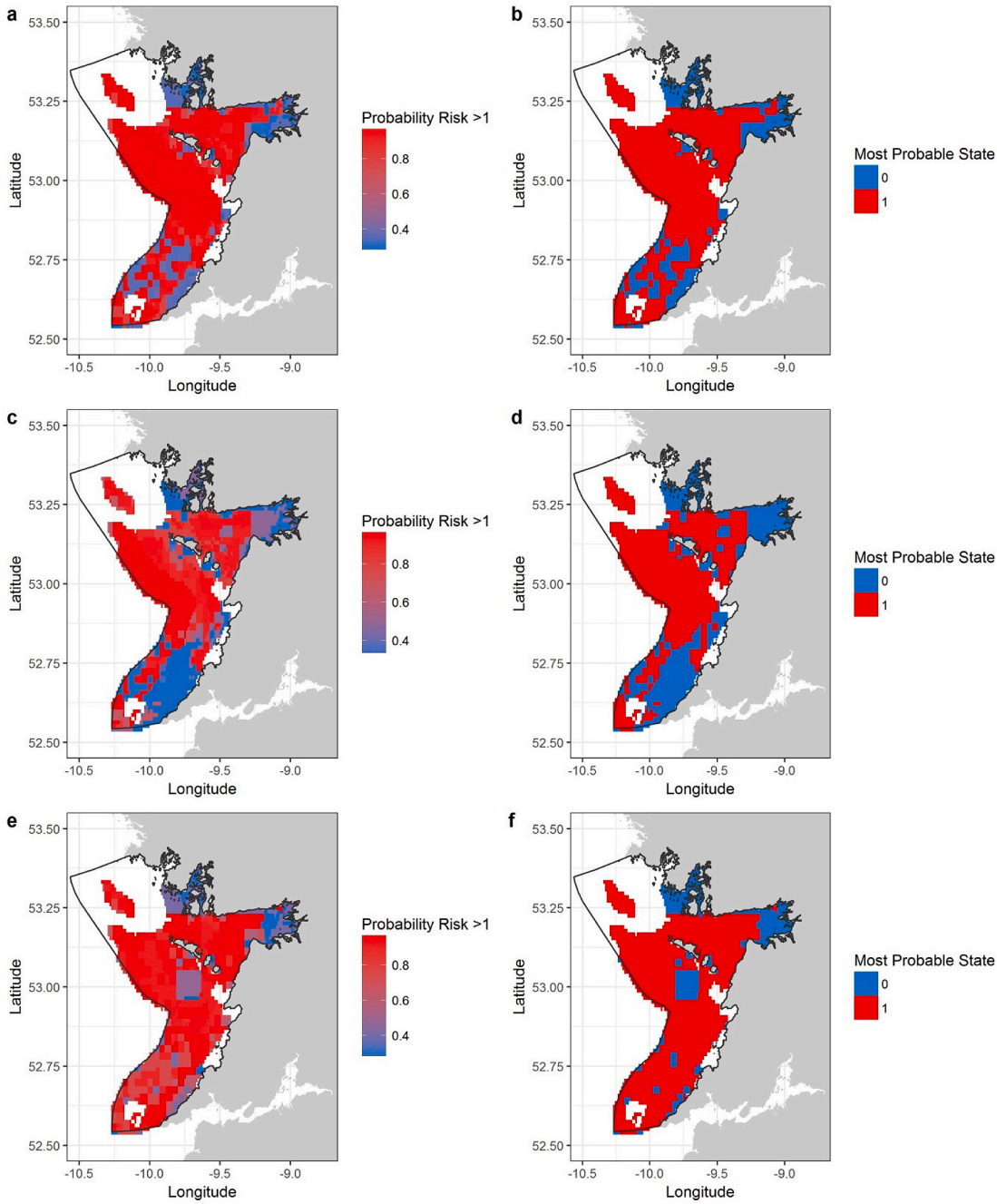


**Fig. 6.** Geonetica outputs showing the probability of each cell being in the high risk category for the business as usual scenario (a), the length exclusion scenario (c), and the marine protected areas scenario (e) and the most probable state for the business as usual scenario (b), the length exclusion scenario (d) and the marine protected area scenario (f).

**Table 3**
The landings value, full time equivalent employment figures and additional monetary output summed across the study area for each of the three scenarios tested. All monetary values are rounded to the nearest €million.

| Scenario | Total landings value (€M) | Indirect output (€M) | Employment (Full time equivalent) |
|---|---|---|---|
| Business as usual | 3.2 | 2.5 | 68 |
| Restriction of vessels greater than 18 m length within 6 nm | 2.3 (−0.9) (28%) | 1.8 (−0.7) (28%) | 49 (−19) (28%) |
| Marine protected area network | 3 (−0.17) (5%) | 2.4 (−0.13) (5%) | 65 (3) (5%) |

derived from VMS as described above.

*2.5.1.2. Exclusion of vessels over 18 m in length from within the 6 nm boundary (length scenario).* In 2018, Ireland passed Policy Directive 1/2019 which will excludes all vessels over 18 m in length from fishing within the 6 nm limits beginning in 2020. This scenario replicates this measure by removing fishing effort within the 6 nm limit for the otter and seine trawlers over 18 m and distributing them among remaining fished grid cells between 6 and 12 nm reflecting the existing spatial pattern of fishing and assuming that catch per unit effort stays the same (Fig. 4a).

*2.5.1.3. Development of a network of marine protected areas (MPA scenario).* Marxan is a decision support tool aimed at designing marine protected area networks based on the principle of achieving conservation objectives or targets at minimum cost (Game et al., 2008). Marxan was used to design a network of MPAs within the study area which protected 100% of the most sensitive habitats, i.e. maerl and sea grasses and 10% of all other habitats. Marxan files were established and run according to standard Marxan procedures as explained in Game et al. (2008) (For further details see supplementary material A). The 'best' Marxan solution, which minimizes overall costs while meeting all afore-mentioned conservation objectives, was used for this scenario. All fishing effort from all four fleets was removed from the MPA network and distributed among remaining fished grid cells in proportion to the existing spatial pattern of fishing and assuming that catch per unit effort stays the same (Fig. 4b).

Using Netica, we calculated the fishing inputs in the Bayesian network model for each given scenario and recorded the resulting change in ecological risk and landings value. The indirect output and FTEs were then calculated using the multipliers as in section 2.4 and summed across the study area. The final Netica models for each of the three scenarios are available in the supplementary material B, C and D. We used the Netica extension, Geonetica, to map the probability values of the scenario outputs for each grid cell in the study area. Maps were produced with outputs showing the most probable ecological risk state for each grid cell and the probability of risk being greater than 1 for each grid cell therefore indicating where mortality exceeds recovery.

A full process flow diagram describing inputs, methods and outputs is shown in Fig. 5.

## 3. Results

Habitats within the study area are predominately mud (29.1%), rock (19.9%) and mixed sediments (19%). The sensitive habitats such as seagrasses and maerl were present in approximately 4% of grid cells despite being prioritised when assigning habitats to grid cells. This highlights the small patchy nature of such habitats in the inshore area (Fig. 1). Otter trawling vessels, over 18 m, (OTB>18 m) have the highest number of hours fished in the study area which is reflected in the frequency swept and landings value for this fleet (Fig. 2).

### *3.1. Business as usual (BAU scenario)*

In the business as usual scenario, 85.3% of the area has a risk greater than 1 (Fig. 6 and supplementary material B), indicating that the majority of the Galway Bay area currently has higher mortality from abrasion in benthic sediments than its ability to recover. The mapped outputs (Fig. 6a and b) indicate that the inshore areas, (where maerl and seagrass habitats dominate) are more likely to be less than 1. The commercially important fishing grounds are in the high risk state and the south of the study area, outside of the Nephrops ground, has the most variability in risk (Fig. 6a and b). The summed annual landings value from fisheries in the study area was €3.2 million. Fisheries in the area equates to an additional €2.3 million in indirect output to the economy and 68 full time equivalent jobs (Table 3).

### *3.2. Exclusion of vessels over 18 m in length from within the 6 nm boundary (length scenario)*

This scenario reduced the number of high risk cells from 85.3% to 70.7%, (Fig. 6 and supplementary material C). The spatial distribution of risk (Fig. 6c and d) shows that reducing fishing of large vessels in the inshore area creates areas, particularly south of the study area, which reduces risk from high to low (i.e. risk score greater than 1 to a risk score less than 1). The landings value assumed to be re-distributed to waters outside 6 nm by this measure is approximately €0.9 million and an additional €0.7 million in indirect output to the economy and 19 full time equivalent employees (Table 3). This represents 28% of the value represented in the business as usual scenario.

### *3.3. Development of a network of MPAs (MPA scenario)*

The Marxan solution included four MPAs in the network covering a total of 504 $km^2$ (Fig. 3b). This scenario reduced ecological risk by just 1.7% to 83.6% (Fig. 6, supplementary material D). This scenario impacted all four fishing fleets with most showing a 4–5% increase in the number of cells with no fishing effort (supplementary material D). The distribution maps (Fig. 6e and f) show that whilst closed areas are exclusively low risk, many of the cells to which fishing was displaced, particularly in the south of the study area, change from low to high risk. The landings value, indirect output and employment which is re-distributed to areas outside the network is approximately €0.17 million, € 0.13 million and 3 FTE's respectively, 5% of the value in the business as usual scenario (Table 3).

## 4. Discussion

In this study we present a method which provides a quantitative, spatial, risk and economic impact assessment of the effects of human activities and pressures on the marine environment. In addition, we assess potential costs and benefits of management options to mitigate against these pressures. Expanding on the existing literature, this study develops a Bayesian network model where both ecological (habitat, pressures, mortality, recovery) and economic (direct and indirect value, employment) data are used to model the probabilities of spatially explicit outcomes from a range of management strategies. The method is applicable to different areas, scales, pressures and ecosystem components and has potential for further development (e.g. comparing multiple human activities and pressures) to aid in the decision making process.

The results show that currently the risk of adverse effects from abrasion was high for 85.3% of the study region. This was unsurprising given the valuable trawling fisheries in the area which is the main contributor to abrasion. However, the use of VMS for estimating fishing effort has several limitations: 1. VMS does not currently account for vessels under 12 m in length which are not required to have VMS onboard. The inability to spatially quantify the activity of the under 12 m fleet is common in fisheries studies and several studies have attempted to map these vessels using other methods (des Clers et al., 2008; Breen et al., 2015). Inclusion of VMS on smaller inshore vessels or other improvement in data acquisition on smaller fleets would greatly improve the accuracy of such assessments. However, in our study area most vessels under 12 m are using static gears which does not contribute greatly to total abrasion pressure. 2. The 2-h time interval between VMS pings may result in a local overestimation of the fished footprint, and therefore the frequency swept, as the whole abrasion pressure is allocated to the cell containing the ping, and conversely in an underestimation of the pressure exerted on adjacent cells. This would mostly affect areas with low fishing pressure, by generating artificially high variability where the abrasion pressure estimate relies on few fishing events and conceals areas where fishing effort may already be low. This would lead to an overestimation of risk. Higher ping frequency would enable abrasion to be mapped on a higher resolution spatial grid and therefore enable better identification of pressure hotspots. The use of frequency swept in cells which intersect the coastline and therefore have a cell size less than 1 $km^2$ may also result in an overestimation of risk, although we believe that this overestimation is minimal given the limited amount of fishing effort from otter trawls and seine nets in cells which intersect with the coastline.

The scenario whereby all vessels over 18 m in length are excluded from within the 6 nm limit showed the largest (14.6%) reduction in the ecological risk from abrasion in the study area. The southern part of the study area was most affected. Here much of the area changed from high

risk to low risk due to the low, yet damaging, amount of fishing in these cells prior to regulation change. This indicates that spatial restrictions which only target a portion of the total fishing effort will benefit areas where fishing effort was already low. If decision makers aimed to reduce overall risk in the area to below 50% of the total area, it would require the complete exclusion of vessels within 6 nm (i.e. reducing risk to zero), rather than restricting only vessels over 18 m in length. The effect of chronic (high and prolonged) levels of fishing on the benthic habitats has been found to shift habitats away from sessile, emergent, high biomass species towards smaller in-faunal species at a lower trophic level (Kaiser et al., 2000; Jennings et al., 2001). This habitat alteration in heavily fished areas means that attention might be best focused on improving these lighter fished areas where the potential for recovery is highest.

The risk assessment method used in this paper involves a number of assumptions. First, in order to gain information on habitats we combined data on sediment types based on estimated relative proportions of each sediment per habitat. This assumes that mortality and recovery per habitat can be related to the portion of sediment per habitat. To remove this assumption we would require improved data on benthic habitats and accurate percentages of sediment per habitat in the area. This would allow us to use recently published meta-analysis such as Sciberras et al. (2018) and Hiddink et al. (2017) that model benthic impacts from trawling given the percentage of sediment per habitat. Second, the recovery equations do not account for the temporal aspect of recovery; in particular a logistic population growth during the recovery phase which would be time (seasonally) dependent. Pitcher et al. (2017) and Hiddink et al. (2019) derive a Relative Benthic Status score using logistic recovery rates which are ecologically intuitive and could provide risk scores if the parameters of the logistic model can be estimated and if time of year of impact, which would influence the recovery rate, could be incorporated. Finally, using an annual value for recovery rates, does not take into consideration the temporal nature of fisheries, particularly of small fleets. This masks the potential of temporal management measures such as seasonal closures when recovery could occur. If the risk assessment model was discretised to smaller time steps it would provide scope for identifying periods of recovery from, for instance, seasonal fisheries and enable comparison of a broader range of management options.

The length category exclusion scenario produced the largest potential effects on the economic factors: landings value, indirect output and full time equivalent employees. These 'potential effects' do not necessarily mean that this value is lost from the system, rather it is the potential value at which the original cost structure is likely to be affected by the spatial closure. When developing scenarios for this study we assumed that fishers moved out of the area, fished for the same total number of hours with the same catch per unit effort and did not change their spatial pattern of fishing outside the area. Whilst it was beyond the scope of this paper to include changes in fisher behaviour within the scenario development it should be an important consideration for decision makers. Modelling fisher behaviour before, during and after spatial closures has revealed that factors such as diesel prices, distance to port and increased competition for fishing grounds could all have an effect on the economic structure post closure (e Costa et al., 2013; Stevenson et al., 2013; Tidd et al., 2015; Wallace et al., 2015). In our length category exclusion scenario, potential new fishing grounds would open up within 6 nm for the vessels under 18 m. Catch and costs per unit effort for these fleets would depend on the total fishing effort that developed following exclusion of vessels over 18 m. For the over 18 m fleet, exclusion from inshore waters might result in 'fishing the line', where fishers increase their effort along the boundary of the spatial closure and which would increase competition in this area (Stevenson et al., 2013). To avoid such competition, fishers might be forced to travel further from home ports to achieve the same catch (Stevenson et al., 2013), this will increase costs and have a knock on effect to the overall catch per unit effort. Incorporating some of these concepts into the scenario development would give a more realistic idea of actual economic impact.

One of the limitations of the economic analysis was the use of national multipliers for calculating indirect output and FTE. These national multipliers are calculated from the Input Output model developed by Grealis and O'Donoghue (2015). The indirect output multiplier for example indicates the additional spending within the national economy from a €1 increase in landings (Grealis et al., 2017). Similarly, the full time equivalent is the number of employees a €1 landings value will generate within the national economy. The use of national multipliers, given the particular characteristics of the fisheries in the area, means that we cannot say where the indirect impacts of the landings will be felt in the wider economy. They only indicate what the total indirect output and employment effects might be at the national level. The national scale multipliers, therefore, are not spatially referenced or linked to locations of fisheries and ports. Future work to generate local level multipliers and in understanding the flow of landings from sea to land would allow for a spatial analysis of economic costs and benefits in the local economy.

The MPA scenario which protected 10% of all habitats and 100% of the two sensitive habitats, showed a decrease in risk within the study area of just 1.7%. Three of the four MPAs overlap at least partially with already designated SACs. The implementation of these closures within the study area, therefore, is a realistic management option. The most interesting finding in this scenario is the localised impact of MPAs on the risk of abrasion. Areas within the MPA moved from high risk to low risk as would be expected, however, other areas which were previously low risk moved to high risk due to displacement of fishers from the study area assuming they distribute their fishing effort in proportion to the existing fishing effort in those areas. Other studies have shown similar effects of MPAs for example, Greenstreet et al. (2009) and Campbell et al. (2014). Greenstreet et al. (2009) found that introducing MPAs to protect ground-fish created an increase in pressure on benthic invertebrates due to displacement. Whilst MPA closures might meet targets for specific sensitive habitats or species and have added benefits for conservation specific objectives they may not be suitable for the ecosystem-based approach to fisheries management nor for meeting conservation targets which aim to protect percentages of the sea floor affected by trawling (e.g. MSFD Descriptor 6 for seafloor integrity targets).

Other types of management measures have been implemented in fisheries management to mitigate abrasion, such as technical changes to gears to reduce abrasion (Valdemarsen et al., 2007). These non-spatial measures could also be compared in the Bayesian network by varying the frequency swept parameter depending upon the gear configurations. In other studies, such as Stelzenmuller et al. (2010, 2015), it was combined with GIS to produce spatially resolved assessments of management measures. Here, we added a decision support tool, Marxan, to aid in the development of scenarios and utilised economic data as well as environmental data to enable a multidisciplinary approach to decision making. The potential flexibility of Bayesian network analysis makes it a powerful tool to compare the effects of management decisions and its future potential in marine management needs further development. For example, in this study we only assessed risk of adverse effects of abrasion from fisheries however managers are often dealing with multiple pressures from multiple activities. Decision support tools such as Marxan with Zones (Watts et al., 2008) allow the development of more complex scenarios including economic costs and benefits which could be used as inputs to more complex Bayesian network models. Comparing between abrasion and other pressures from various activities such as smothering from aquaculture or bycatch from static fishing gears is possible with the Bayesian Network. Other studies have developed methods of assessing bycatch within fisheries (Brown et al., 2015; Breen et al., 2017). However, suitable methods to assess the risk of adverse effects from other pressures such as smothering are yet to be developed. Future work should focus on the development of risk assessment methods for other

activities and integrate multiple pressures into a Bayesian network which would provide decision support for the implementation of an ecosystem approach to marine management.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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## Appendix A. Supplementary data

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