Document Control

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Southern Inshore Fisheries and Conservation Authority (IFCA)
Marine Conservation Zone Fisheries Assessment (Part B)

Marine Conservation Zone: Studland

Feature: Seagrass Beds, Subtidal Sand

Broad Gear Type: Bottom Towed Fishing Gear

Gear type(s) Assessed: Beam trawl / Light Otter Trawl and Pump Scoop Dredges

Technical Summary

As part of the MCZ assessment process for the tranche three Studland MCZ, it was identified that trawling (specifically light otter trawl & beam trawl) and dredges (specifically clam/pump scoop) and their potential impacts required an in-depth assessment. Fishing activity is the site is very low, with currently no known vessels actively trawling or dredging in the site.

The potential pressures likely to be exerted by the activity upon designated features were identified as abrasion, disturbance and penetration of the seabed below and on the surface and the removal of non-target and target species.

Scientific literature shows that whilst trawling has the potential to cause physical and biological disturbance, the extent and severity of impact largely depends on site-specific factors including sediment type and physical regime. As such, the level of impact can largely vary between studies conducted in 'similar' habitat types. Studies into the effects of trawling and dredging over seagrass beds showed that the gear can lead to the direct and immediate removal of the feature.

When considering that no vessels are known to actively fish in the Studland MCZ, in combination with other evidence (scientific literature, sightings data, feature mapping) and site-specific factors, it was concluded the activity is not likely to pose a significant risk to subtidal sand feature. Therefore, current management is considered sufficient to protect subtidal sand from all types of bottom towed fishing gear. This was concluded due to the current level of activity (none) and the low sensitivity of the feature. As such, it is believed the activity will not hinder the achievement of the designated feature to achieve its 'maintain' general management approach and that it is compatible with the site's conservation objectives. Existing management measures are therefore considered sufficient to ensure that trawling and scallop dredging remains consistent with the conservative objectives of the site, fishing effort will continue to be monitored.

When considering the above it was concluded that trawling and dredging was likely to pose a significant risk to the seagrass beds feature. One instance of interaction of the gear with the feature could lead to the immediate removal of the seagrass itself and recovery is slow. Therefore, additional management which protects seagrass beds from all types of bottom towed fishing gear will be created covering the seagrass beds feature. As such, it is believed the activities will not hinder the achievement of the designated features 'recover' general management approach and that it is compatible with the site's conservation objectives. Existing and proposed management measures are therefore considered sufficient to ensure that trawling and scallop dredging remains consistent with the conservative objectives of the site, fishing effort will continue to be monitored.

Contents

1 Introduction

1.1 Need for an MCZ assessment

This assessment has been undertaken by Southern IFCA in order to document and determine whether management measures are required to achieve the conservation objectives of Studland Marine Conservation Zone (MCZ). Southern IFCA has duties under section 154 of the Marine and Coastal Access Act 2009 which states;

154 Protection of marine conservation zones

(1) The authority for an IFC district must seek to ensure that the conservation objectives of any MCZ in the district are furthered.

(2) Nothing in section 153(2) is to affect the performance of the duty imposed by this section.

(3) In this section—

(a) “MCZ” means a marine conservation zone designated by an order under section 116;

(b) the reference to the conservation objectives of an MCZ is a reference to the conservation objectives stated for the MCZ under section 117(2)(b).

Section 125 of the 2009 Act also requires that public bodies (which includes the IFCA) exercise its functions in a manner to best further (or, if not possible, least hinder) the conservation objectives for MCZs.

This MCZ assessment will complement Southern IFCA’s assessment of commercial fishing activities in European Marine Sites (EMS) – designated to protect habitats and species in line with the EU Habitats Directive and Birds Directive. To bring fisheries in line with other activities, the Department for Environment, Food and Rural Affairs (DEFRA) announced on the 14th August 2012 a new approach to manage fishing activities within EMSs. This change in approach will promote sustainable fisheries while conserving the marine environment and resources, securing a sustainable future for both.

1.2 Documents reviewed to inform this assessment

- Reference list (Section 7)
- Defra’s matrix of fisheries gear types and European Marine Site protected features[1]
- Site map(s) – feature location and extent (Annex 1)
- Natural England’s Advice on Operations for Studland Bay MCZ[2]
- Natural England’s Supplementary Advice for Studland Bay MCZ[3]
- Fishing activity data (map(s), etc) (Annex 5)
- Fisheries Impact Evidence Database (FIED)

1.3 Overview and designated features

Studland MCZ was designated in May 2019 and covers the bay between Old Harry rocks, Studland and the entrance to Poole Harbour. The site covers an area of approximately 4 km^2 and protects intertidal coarse sediment, seagrass beds and subtidal sand, which supports a range of communities including worms, crustaceans and molluscs. The site also protects the species the long-snouted seahorse (*Hippocampus guttulatus*).

A summary of the site’s designated features is provided in Table 1, together with the recommended General Management Approach (GMA) for each feature. The GMA required for a feature in a MCZ will either be for it

[1] https://www.gov.uk/government/publications/fisheries-in-european-marine-sites-matrix

[2] https://designatedsites.naturalengland.org.uk/Marine/FAPMatrix.aspx?SiteCode=UKMCZ0072&SiteName=studland&SiteNameDisplay=Studland+Bay+MCZ&countyCode=&responsiblePerson=&SeaArea=&IFCAArea=&NumMarineSeasonality=,0

[3] https://designatedsites.naturalengland.org.uk/Marine/SupAdvice.aspx?SiteCode=UKMCZ0072&SiteName=studland&SiteNameDisplay=Studland+Bay+MCZ&countyCode=&responsiblePerson=&SeaArea=&IFCAArea=&NumMarineSeasonality=,0,0

to be maintained in favourable condition (if it is currently in this state), or for it to be recovered to favourable condition (if it is currently in a damaged state) and then to be maintained in favourable condition.

Table 1.Designated features and General Management Approach

Designated Feature	General management approach
Intertidal coarse sediment	Maintain in favourable condition
Subtidal Sand	Maintain in favourable condition
Long-snouted seahorse (*Hippocampus guttulatus*)	Maintain in favourable condition
Seagrass beds	Recover to favourable condition

Please refer to Annex 1 for site feature maps of broad-scale habitats and features of conservation importance. This feature data comes from the Natural England, 2019 data set given to Southern IFCA, containing a collation of marine habitat and species records that contribute to the designation of marine habitats and features. This corresponds with the feature data on Magic Map which represents Natural England's best available evidence (https://magic.defra.gov.uk/MagicMap.aspx).

1.4 Conservation objectives

The site's conservation objectives apply to the Marine Conservation Zone and the individual species and/or habitat for which the site has been designated (the "Designated features" listed below).

The conservation objective of each of the zones is that the protected habitats:

1. are maintained in favourable condition if they are already in favourable condition
2. be brought into favourable condition if they are not already in favourable condition

For each protected feature, favourable condition means that, within a zone:

1. its extent is stable or increasing
2. its structure and functions, its quality, and the composition of its characteristic biological communities (including diversity and abundance of species forming part or inhabiting the habitat) are sufficient to ensure that its condition remains healthy and does not deteriorate

Any temporary deterioration in condition is to be disregarded if the habitat is sufficiently healthy and resilient to enable its recovery.

For each species of marine fauna, favourable condition means that the population within a zone is supported in numbers which enable it to thrive, by maintaining:

1. the quality and quantity of its habitat
2. the number, age and sex ratio of its population. Any temporary reduction of numbers of a species is to be disregarded if the population is sufficiently thriving and resilient to enable its recovery.

Any alteration to a feature brought about entirely by natural processes is to be disregarded when determining whether a protected feature is in favourable condition.

2 MCZ assessment process

2.1 Overview of the assessment process

The assessment of commercial fishing activities within the Studland MCZ will be undertaken using a staged process, akin to that proposed by the Marine Management Organisation (MMO)[4], for marine license applications (Annex 2). The assessment process comprises of an initial screening stage to establish whether an activity occurs or is anticipated to occur/has the potential to occur within the site. Activities which are not screened out are subject to a simple 'part A' assessment, akin to the Test of Likely Significant Effect required by article 6(3) of the Habitats Directive. The aim of this assessment is to identify pressures capable of significantly affecting designated features or their related processes. Fishing activities and their associated pressures which are not screened out in the part A assessment and then subject to a more detailed 'part B' assessment, where assessment is undertaken on a gear type basis. A part B assessment is akin to the Appropriate Assessment required by article 6(3) of the Habitats Directive. The aim of this assessment is to determine whether there is a significant risk of the activity hindering the conservation objectives of the MCZ. Within this stage of assessment, 'hinder' is defined as any act that could, either alone or in combination:

- in the case of a conservation objective of 'maintain', increase the likelihood that the current status of a feature would go downwards (e.g. from favourable to degraded) either immediately or in the future (i.e. they would be placed on a downward trend); or

- in the case of a conservation objective of 'recover', decrease the likelihood that the current status of a feature could move upwards (e.g. from degraded to favourable) either immediately or in the future (i.e. they would be placed on a flat or downward trend) (MMO, 2013).

If the part B assessment is unable to conclude that there is no significant risk of an activity hindering the conservation objectives of the MCZ, then the activity may be subject to management and consideration will be given to whether or not the public benefit of the activity outweighs the risk of damage to the environment; and if so, whether the activity is able to deliver measures of equivalent environmental benefit to the damage that is likely to occur to the MCZ.

2.2 Screening and part A assessment

The aim of the screening stage and part A assessment is to determine whether, under section 125 and 154 of MCAA, fishing activities occurring or those which have the potential to occur within the site are compatible with the conservation objectives of the MCZ.

The screening of commercial fishing activities in Studland MCZ was undertaken using broad gear type categories. Sightings data collected by the Southern IFCA, together with officers' knowledge, was used to ascertain whether each activity occurs within the site, or has the potential to occur/is anticipated to occur in the foreseeable future. For these occurring/potentially occurring activities, an assessment of pressures upon MCZ designated features was undertaken using Natural England's Advice on Operations for the Feature (using an alternate designated site as the Conservation Advice for the Studland MCZ has not yet been produced.

Activities were screened out for further part B assessment if they satisfied one or more of the following criteria:

1. The activity does not occur within the site, does not have the potential to occur and/or is not anticipated to occur in the foreseeable future.

2. The activity does occur but the pressure(s) does not significantly affect/ interact with the designated feature(s).

4 https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/410273/Marine_conservation_zones_and_marine_licensing.pdf

3. The activity does occur but the designated feature(s) is not sensitive to the pressure(s) exerted by the activity.

2.3 Screening of commercial fishing activities based on occurrence

Initial screening was undertaken to identify the commercial fishing activities which currently occur within the site, together with those which have the potential to occur or/and are reasonably foreseen to occur in the future (Annex 3). To maintain consistency with Southern IFCA's assessment of commercial fishing activities in European Marine Sites, the individual gear types identified in Defra's matrix were assessed and these were grouped into broad gear types.

2.4 Screening of commercial fishing activities based on pressure-feature interaction

Fishing activities which were identified as occurring, have the potential to occur and/or are anticipated to occur in the foreseeable future within the site were screened with respect to the potential pressures which they may be exert upon designated features (Part A assessment). This screening exercise was undertaken using Natural England's Advice on Operations (Annex 4) and Supplementary Advice for Studland Bay MCZ. The Advice on Operations provides a broad scale assessment of the sensitivity of designated features to different activity-derived pressures, using nationally available evidence on their resilience (an ability to recover) and resistance (the level of tolerance) to physical, chemical and biological pressures. The assessments of sensitivity to these pressures are measured against a benchmark. It should be noted that these benchmarks are representative of the likely intensity of a pressure caused by typical activities, and do not represent a threshold of an 'acceptable' intensity of a pressure. It is therefore necessary to consider how the level of fishing intensity observed within Studland MCZ compares with these benchmarks when screening individual activities.

Due to the broad-scale nature of the sensitivity assessments provided in Natural England's Advice on operations, each pressure is assigned a risk profile based upon the likelihood of the pressure occurring and the magnitude of the impact should that pressure occur. These risk profiles have been used, together with site-specific knowledge, to identify those pressures which could significantly affect designated features.

The Natural England Advice on Operations for Studland Bay MCZs used is provided in Annex 4. The resultant activity pressure-feature interactions which have been screened in for bottom towed fishing gear for the part B assessment are summarised in Tables 2,3 & 4 for sensitive designated features. The activity pressure-feature interactions which were screened out in the Part A Assessment are detailed in a standalone document ('Screening and Part A Assessment') for The Studland MCZ.

Table 2. Summary of fishing pressure-feature screening for Subtidal Sand and dredges/demersal trawls. Please note only pressures screened in for the part B assessment are presented here.

Potential Pressures	Sensitivity	Considered in Part B	Justification	Relevant Attributes
Abrasion/disturbance of the substrate on the surface of the seabed	S	Y	This gear type is known to cause abrasion and disturbance to the seabed surface. Further investigation is needed on the magnitude of the pressure including spatial scale/intensity of the activity and location of the activity in relation to the feature.	Distribution: presence and spatial distribution of biological communities; Structure and function: presence and abundance of key structural and influential species; Structure: sediment composition and distribution; Structure: species

				composition of component communities
Changes in suspended solids (water clarity)	S	Y	This gear is known to cause the resuspension of finer sediments therefore further assessment is required.	Supporting processes: water quality - turbidity
Penetration and/or disturbance of the substrate below the surface of the seabed, including abrasion	S	Y	This gear type is known to cause penetration and disturbance to the seabed surface. Further investigation is needed on the magnitude of the pressure including spatial scale/intensity of the activity and location of the activity in relation to the feature.	Distribution: presence and spatial distribution of biological communities; Structure and function: presence and abundance of key structural and influential species; Structure: sediment composition and distribution; Structure: species composition of component communities
Removal of non-target species	S	Y	Impacts on the associated community may occur through the removal of larger epifaunal and potentially infaunal species, whilst smaller organisms are likely to pass through the gear. Abrasion, resulting from contact with the gear, however is likely to disturb smaller species. There is no site-specific information on the communities associated with this feature as it is newly designated. General information on the designated features from the MCZ features catalogue provides a general description. The communities associated with subtidal sand depend on the level of silt/clay content. In sheltered areas, different animals will be found depending on the sand/mud ratio with flat fish, sand eels, worms and bivalves present on open coasts. Further investigation is needed as to the magnitude of	Distribution: presence and spatial distribution of biological communities; Structure and function: presence and abundance of key structural and influential species; Structure: sediment composition and distribution; Structure: species composition of component communities

			disturbance to associated communities/species and location of the activity in relation to the feature.	
Removal of target species	NA	Y	Impacts on the associated community may occur through the removal of larger epifaunal and potentially infaunal species, whilst smaller organisms are likely to pass through the gear. Abrasion, resulting from contact with the gear, however is likely to disturb smaller species. There is no site-specific information on the communities associated with this feature as it is newly designated. General information on the designated features from the MCZ features catalogue provides a general description. The communities associated with subtidal sand depend on the level of silt/clay content. In sheltered areas, different animals will be found depending on the sand/mud ratio with flat fish, sand eels, worms and bivalves present on open coasts. Further investigation is needed as to the magnitude of disturbance to associated communities/species and location of the activity in relation to the feature.	Distribution: presence and spatial distribution of biological communities; Structure and function: presence and abundance of key structural and influential species; Structure: species composition of component communities
Smothering and siltation rate changes (Light)	S	Y	This gear is known to cause the resuspension of finer sediments therefore further assessment is required.	Supporting processes: water quality - turbidity

Table 3. Summary of fishing pressure-feature screening for seagrass beds and dredges/demersal trawls. Please note only pressures screened in for the part B assessment are presented here.

Potential Pressures	Sensitivity	Considered in Part B	Justification	Relevant Attributes
Abrasion/disturbance of the substrate on the surface of the seabed	S	Y	This gear type is known to cause abrasion and disturbance to the seabed surface. Further investigation is needed on the magnitude of the pressure including spatial scale/intensity	Distribution: presence and spatial distribution of biological communities; Extent and distribution; Structure and

			of the activity and location of the activity in relation to the feature.	function: presence and abundance of key structural and influential species; Structure: biomass; Structure: sediment composition and distribution; Structure: species composition of component communities
Changes in suspended solids (water clarity)	S	Y	This gear is known to cause the resuspension of finer sediments therefore further assessment is required.	Supporting processes: light levels; Supporting processes: water quality - turbidity
Penetration and/or disturbance of the substrate below the surface of the seabed, including abrasion	S	Y	This gear type is known to cause penetration and disturbance to the seabed surface. Further investigation is needed on the magnitude of the pressure including spatial scale/intensity of the activity and location of the activity in relation to the feature.	Distribution: presence and spatial distribution of biological communities; Extent and distribution; Structure and function: presence and abundance of key structural and influential species; Structure: biomass; Structure: sediment composition and distribution; Structure: species composition of component communities
Removal of non-target species	S	Y	Impacts on the feature and associated community may occur through the removal of the feature itself, larger epifaunal and potentially infaunal species, whilst smaller organisms are likely to pass through the gear. Abrasion, resulting from contact with the gear, however is likely to disturb smaller species. There is no site-specific information on the communities associated with this feature as it is newly designated. General information on the designated features from	Distribution: presence and spatial distribution of biological communities; Structure and function: presence and abundance of key structural and influential species; Structure: species composition of component communities

			the MCZ features catalogue. Seagrass beds provide nursery habitat for young fish and shellfish, as well as sheltered home for other animals such as pipefish and seahorses. Further investigation is needed as to the magnitude of disturbance to associated communities/species and location of the activity in relation to the feature.	
Smothering and siltation rate changes (Light)	S	Y	This gear is known to cause the resuspension of finer sediments therefore further assessment is required.	Supporting processes: sedimentation rate; Supporting processes: water quality - turbidity

3 Part B Assessment

The aim of the part B assessment is for the IFCA to ensure that that there is no significant risk of a fishing activity hindering the conservation objectives of the MCZ; and to confirm that the authority is able to exercise its functions to further the site's conservation objectives.

In order to adequately assess the potential impacts of an activity upon a designated feature, it is necessary to consider the relevant attributes of that feature that may be affected. Attributes are provided in Natural England's Supplementary Advice on Conservation Objectives (SACOs) and represent the ecological characteristics or requirements of the designated species and habitats within a site. These attributes are considered to be those which best describe the site's ecological integrity and which if safeguarded will enable achievement of the Conservation Objectives. Each attribute has an associated target which identifies the desired state to be achieved; and is either quantified or qualified depending on the available evidence. No Supplementary Advice is currently available for Studland MCZ, therefore after relevant pressures were identified from the pressure-feature interaction screening (part A assessment), suitable attributes were identified from existing Natural England's Supplementary Advice packages for the three alternative MCZs. These are outlined in Table 2, 3 and 4.

3.1 Assessment of Trawling and Dredging in the Studland MCZ

3.1.1 Summary of the Fishery

Trawling can take place all year around in the area surrounding the Studland MCZ. The level of activity is however low with only one vessel able to take part in the fishery with its home port in the area. Fishing currently occurs in Poole Bay, off of Bournemouth. In the site the activity is not known to have occurred. When using this method of fishing the species caught is dependent on the time of year. Catches can include common sole (*Solea solea*) and European plaice (*Pleuronectes platessa*), with a bycatch of bass.

3.1.2 Technical gear specifications

3.1.3 Light otter trawl

An otter trawl comprises of following design (see Figure 1). Two shaped panels of netting are laced together at each side to form an elongated funnel shaped bag (Seafish, 2015). The funnel tapers down to a cod-end where fish are collected (Seafish, 2015). The remaining cut edges of the net and net mouth are strengthened by lacing them to ropes to form 'wings' that are used to drive fish into the net (Seafish, 2015). The upper edge of the rope is referred to as the head line, the lower edge is referred to as the foot rope of fishing line and side ropes are known as wing lines (Seafish, 2015). Floats are attached to the headline to hold the net open and the foot rope is weighted to maintain contact with the seabed and prevent damage to the net (Seafish, 2015). The wings of the net are held open by a pair of trawl doors, also known as otter boards, and are attached to the wings by wires, ropes or chains known as bridles and sweeps (Seafish, 2015). The sweep connects the trawl door to top and bottom bridles which are attached to the headline and footrope of the net, respectively (Seafish, 2015). The choice of material used for the sweeps and bridles depends on the size of gear and nature of the seabed, with smaller inshore boats using thin wire and combination rope (Seafish, 2015). The trawl doors, which are made of wood or steel are towed through the water at an angle which causes them to spread apart and open the net in a horizontal direction (Seafish, 2015). The trawl doors are attached to the fishing vessel using wires referred to as trawl warps (Seafish, 2015). The trawl doors must be heavy enough to keep the net on the seabed as it is towed (Seafish, 2015). As the trawl doors are towed along the seabed they generate a sediment cloud which helps to herd fish towards the mouth of the trawl (Seafish, 2015). The bridles and sweeps continue the herding action of the trawl doors as the trail on the seabed and disturb the sediment, creating a sediment cloud (Seafish, 2015). The length of the sweeps and bridles and distance between the two trawl doors is tuned to the target species (Seafish, 2015). Species such as lemon sole and plaice can be herded into the trawl over long distances and so the length of the sweeps is longer (Seafish, 2015).

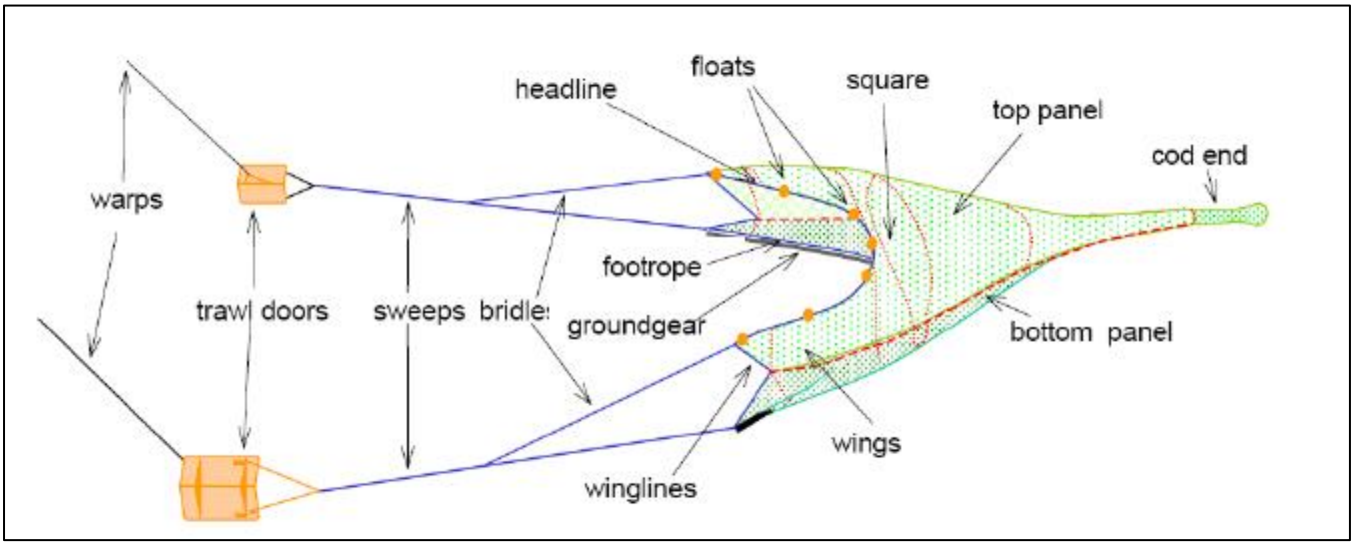


Figure 1. Key components of an otter trawl.

Source: www.seafish.org/upload/b2b/file/r_d/BOTTOM%20TRAWL_5a.pdf

The mesh size of the net used varies depending on the type of trawl (Seafish, 2015). In the UK, there has been a move towards an increase in mesh size, particularly in the top panel and wings, in order to improve gear selectivity (Seafish, 2015).

The ground rope will have some form of ground gear attached to protect the netting from damage on the seabed (Seafish, 2015). The ground gear can largely vary. The most basic is where bare fishing line and the netting is laced directly to the rope of combination rope (Seafish, 2015). Chains may also be used and the style of attachment can vary (Seafish, 2015). Ground gear may also include bobbins and rock hoppers which commonly use small and large rubber discs (up to 600 mm) (Seafish, 2015).

The drag of the gear, combined with the floats on the headline, mean the weight of the trawl on the seabed is in the region of 10 to 20% of what it would be in air (Seafish, 2015).

A light otter trawl is one that uses anything less than the definition given for a heavy otter trawl, which include any of the following (MMO, 2014):

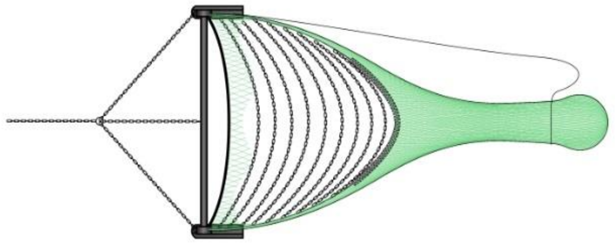
- Sheet netting of greater than 4 mm twine thickness
- Rockhoppers or discs of 200 mm or above in diameter
- A chain for the foot/ground line (instead of wire)

Generally, vessels will shoot and haul their gear over the stern of the boat (Seafish, 2015). Restrictions on vessels over 12 metres in length in the Southern IFCA district limits the size of gear that can be used within the district.

3.1.4 Beam trawl

A net is held open by a rigid framework to maintain trawl opening, regardless of towing speed, in addition to supporting the net (Seafish, 2015). The framework consists of a heavy tubular steel beam which is supported by steel beam heads at each end. Each beam head has wide shoes at the base which slide over the seabed (Seafish, 2015). A cone shaped net is towed from the framework, with the head rope attached to the beam and foot rope connected to the base of the shoes (Seafish, 2015). The footrope forms a 'U' shape curve behind the beam as it is towed over the seabed (Seafish, 2015). The beam is towed using a chain bridle which is attached to both shoes and at the centre of the beam; all coming together to form a single trawl warp which leads to the vessel (Seafish, 2015).

There are two types of beam trawl and these are referred to as 'open gear' and 'chain mat gear' (Seafish, 2015). Open gear uses a lighter rig, with a number of chains, known as 'ticklers', which are towed along the seabed across the mouth of the net (Figure 2) (Seafish, 2015). Tickler chains help to disturb fish from a muddy seabed. Open gear is used on clean and soft ground. Chain mat gear on the other hand is used for towing over harder and stonier seabed and if often used by larger vessels (Seafish, 2015). The chain mat gear uses a lattice work of chains which are towed from the back of the beam and attach to the footrope of the net (Figure 3) (Seafish, 2015). Lighter styles of beam, using fewer tickler chains and without a chain mat, are used to target shrimp (Seafish, 2015).



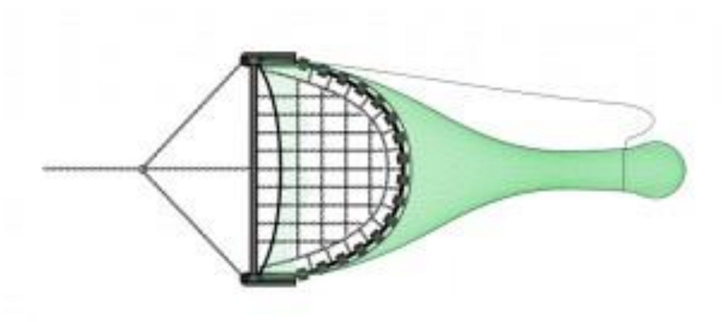


Figure 2. 'Open gear' beam trawl. **Figure 3. 'Chain mat gear' beam trawl.**

Generally, vessels below 12 metres, like those used in the Southern IFCA district, tow one trawl from the stern of the vessel (Seafish, 2015). The size of the beam towed, and the horsepower of many vessels, can be restricted by the local fishery regulations (Seafish, 2015).

3.1.5 Pump Scoop Dredge

Fishing for shellfish in Poole Harbour is carried out using pump-scoop dredge. A pump-scoop dredge consists of toothed dredge basket which is towed through the seabed alongside a vessel (Jensen *et al*., 2005). Attached to the front end of the dredge is a series of water jets which direct a flow of water to the rear of the dredge basket (Jensen *et al*., 2005) (Figure 1). The water jets, powered by a hydraulic pump, allow sediment to be moved through the dredge basket (Jensen *et al*., 2005). In 2012, the use of a trailed pump-scoop dredge, which uses the aid of a davit arm and winch, was introduced. This type of dredge evolved from the previously used and more physically demanding hand-held dredge or scoop, pushed into the sediment and pulled along by a vessel (Jensen *et al*., 2005; Clarke *et al*., 2018). The pump-scoop dredge is deployed from small (less than 10 metre in length) and shallow drafted vessels. This gear type is unique to Poole Harbour

and differs from suction or hydraulic dredging techniques which both fluidise the sediment by spraying water in front of the dredge (Jensen *et al*., 2005).

Figure 4. Typical pump-scoop dredge set up with basket dredge, water jets, davit arm and sorting riddle.

A comparison between the pump-scoop and hand-held dredge revealed no differences in the areas fished in terms of proximity to the shore (i.e. potential displacement of birds) or sediment penetration (i.e. likelihood of impacting on infaunal communities). Further observations also showed no increase in fishing intensity when comparing both dredge types.

The pump-scoop dredge is towed in a circular motion with each tow lasting from 2 to 5 minutes depending on the nature of the seabed. After each tow the pump-scoop dredge is lifted into the vessel and the contents of the dredge basket are emptied directly onto the riddle for sorting. Fishers must sort their catch immediately and return all shellfish under minimum size restrictions, as well as bycatch, to the water.

The configuration of the pump-scoop dredge is dictated by the conditions of the permit. These include restrictions on the dimensions of a dredge basket to a maximum of 460 mm in width, 460 mm in depth and 30 mm in height (excluding any poles or attachment). Dredges must be constructed on rigid bars having spaces of no less than 18 mm between them. Bar spacing is designed to allow young spat and infauna to go through the dredge basket (Jensen *et al*., 2005). A riddle with bar spacing of 18 mm is mandatory for the sorting of shellfish.

3.1.6 Location, Effort and Scale of Fishing Activities

Light otter trawling takes place subtidally and is not known to occur inside the site. The number of vessels engaged in the activity is limited to 1 vessel in the area which operates out of Poole Harbour. This vessel has historically used light otter trawls in Poole Bay.

Beam Trawling takes place subtidally and is not known to occur in the site. However, there are a small number of vessels which have been known to beam trawl in Poole Bay.

Up to 45 vessels partake in the Poole harbour pump scoop clam and cockle fisheries. However, none are known to fish out side of the harbour. A couple of these vessels are known to have ‘trialled’ the pump scoop method in the northern end of Studland bay, however it proved unsuccessful and they have not returned to the site since.

Based on the information described above; trawling and pump scoop fishing do not occur in the Studland Bay MCZ. Hall *et al.* (2008) assessed the sensitivity of marine habitats and species to fishing activities. According to their fishing intensity categories[5] the fishing level in Yarmouth to Cowes MCZ is classed as a Single Pass (Single pass of fishing activity in a year overall).

No sightings of fishing vessels actively fishing in Studland Bay have been made over the past 11 years.

3.2 Seagrass Beds – *Zostera marina*

Z. marina is a salt water flowering plant which resembles terrestrial grass in appearance. It grows seasonally (spring and summer) governed by environmental parameters such as light, nutrients and temperature. Optimum growth temperature is between 10 and 20°C (Nejrup and Pedersen, 2008). Shoots of *Z. marina* are anchored into the sediment via a network of horizontal rhizomes and roots. These rhizomes produce a mat which expands horizontally and can produce further shoots.

Seagrass beds are considered to be one of the most productive of shallow sedimentary marine habitats. The complex nature of the shoots, rhizomes and roots provides habitat for a wide range of flora and fauna. The leaves and shoots themselves provide substrate for algae and anemones, whilst the space between shoots provide nursery habitat for a range of fish (including seahorses), crustaceans, amphipods and cephalopods (Davison and Hughes, 1998).

Seagrass in Studland bay is spread throughout the bay with the largest beds along the sheltered southern half, with smaller patches in the northern areas. The site is known to support immature commercial species including pollack, wrasse, cuttlefish and the common cockle as well as many other crustaceans, molluscs, polychaetes and cnidaria (Seastar Survey Ltd., 2012).

3.3 Pressures

3.3.1 Abrasion/disturbance of the substrate on the surface of the seabed / Penetration and/or disturbance of the substrate below the surface of the seabed, including abrasion.

Abrasion and disturbance is generally related to the direct and physical effects of bottom towed fishing gear. Such effects include the scraping and ploughing of the substrate, scouring and flattening of the seabed, sediment resuspension and changes in the vertical redistribution of sediment layers (Roberts *et al*. 2010).

Otter trawl

Otter trawl fishing gear has contact with the seabed through the ground rope, chains and bobbins, sweeps, doors and any chaffing mats or parts of the net bag (Jones, 1992). Otter boards, or doors, leave distinct tracks on the seafloor ploughing distinct groove or furrows, which can be 0.2-2 metres wide and up to 30 centimetres deep (Jones, 1992; Thrush & Dayton, 2002). The depth of furrows depends on the weight of the board, the angle of attack, towing speed, and the nature of the substrate, being greatest in soft mud (Jones, 1992; Løkkeborg, 2005). The passage of the doors also creates sediment mounds known as berms (Gilkinson *et al*. 1998; Johnson *et al*. 2002). Marks on the seabed caused by other parts of the gear are faint when compared with those caused by trawl doors (Løkkeborg *et al*. 2005). Ground ropes and weights can scour and flatten the seabed, skimming the surface sediment between the grooves left by the trawl doors (Jones, 1992; Roberts *et al*. 2010; Grieve *et al*., 2014). Spherical footrope bobbins can cause compressed tracks on surficial sediments (Brylinsky *et al*. 1994). In areas of surface roughness i.e. sand waves and ripples, features can be flattened and the habitat smoothed (Kaiser & Spencer, 1996; Tuck *et al*., 1998; Schwinghamer *et al*., 1996; 1998). It has been reported that the bridles do not appear to result in any marks on the seabed (Brylinsky *et al*. 1994).

Experimental flounder trawling, using an 18 m trawl with 200 kg doors and footrope with 29 cm rubber rollers, in the Bay of Fundy revealed that trawl doors made furrows that were 30 – 85 cm wide and up to 5 cm deep

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[5] Heavy – Daily in 2.5 nm x 2.5 nm, Moderate – 1 to 2 times a week in 2.5 nm x 2.5 nm, Light – 1 to 2 times a month during a season in 2.5 nm x 2.5 nm, Single pass – Single pass of fishing activity in a year overall

in an intertidal area characterised by silty sediments (Brylinsky *et al*. 1994). The same study reported an area of approximately 12% between the outer edges of the doors was visually disturbed (Brylinsky *et al*. 1994). A side-scan survey, used to assess the effects of otter trawl over sand and mud sediments in lower Narragansett Bay, revealed 5 to 10 cm deep tracks from otter trawl doors and 10 to 20 cm high berms in mud bottom channels (DeAlteris *et al*., 1999). No information on the type of gear used was provided in the study. Sediment profile images (SPIs) were used to estimate the physical impacts of experimental trawling using a shrimp otter trawl with a head rope length of 10 m, otter boards measuring 90 x 140 cm and weighing 125 kg each and ground rope of 14 m with 20 kg of lead weight distributed across its length in an area of muddy sediments in the Gullmarfjord (Nilsson & Rosenberg, 2003). Forty-three percentage of the images in trawl area had signs of physical disturbance (Nilsson & Rosenberg, 2003). A crude estimate of the scale of disturbance was made from the images, with an estimated depth of the trawl tracks at approximately 10 cm, and width between 30 and 60 cm (Nilsson & Rosenberg, 2003). It was calculated that one-tenth of the area affected by trawling would be directly affected by ploughing from the otter boards themselves (Nilsson & Rosenberg, 2003).

Beam trawl

The gear used by beam trawl is known to penetrate the seabed, leaving tracks and disturbing the surface sediments (Gubbay & Knapman, 1999). Beam trawls flatten seabed features and can also leave trenches in soft sediment (Tuck *et al*., 1998).It is important to point out however that generally speaking beam trawling does not occur in mud habitats as it cannot be used effectively in such habitat types (Kaiser *et al*. 2002). Studies have revealed that the penetration depth of tickler chains on a beam trawl range from a few centimetres to at least 8 cm (Løkkeborg, 2005). Using a light beam trawl, of 700 kg with 15 tickler chains, disturbance was revealed to be restricted to the upper 1 cm in sandy sediments and 3 cm in muddy silt (Bridger, 1972). An average penetration depth of 40 to 70 mm was reported by de Groot *et al*. (1995). Experimental trawling, using a 3.5 tonne 4 m beam trawl with chain matrix, led to the flattening of sand ripples, suspension of fine materials and a reduction in the consolidation of sediments in areas of stable coarse sand and gravel and mobile sand in the eastern Irish sea (Kaiser & Spencer 1996, Kaiser *et al*. 1996, 1998, 1999). In the North Sea, experimental trawling, using a 7000 kg 12 m beam trawl with tickler chains, resulted in the physical penetration of the year to at least 6 cm in an area of medium hard sandy sediment (Bergman *et al*. 1990; Bergman & Hup, 1992).

Shellfish dredges

There are a number of ways in which mechanical shellfish dredges can cause physical disturbance and these include an increase in sediment suspension above background levels, an increase in turbidity as a result of resuspension, the creation of sediment plumes and a change in sediment composition (Mercaldo-Allen & Goldberg, 2011; Wheeler *et al*., 2014). The most obvious form of physical disturbance are changes in topography (Natural England, 2014). Typically impacts include the creation of depressions and trenches and the smoothing of ripples or creation of ridges within sand environments (Wheeler *et al*., 2014). Intertidal shellfish dredging can result in furrows up to tens of centimetres deep (Kaiser *et al*., 2006). The depth and width of a trench is largely determined by the mode of fishing, gear type and target species (Wheeler *et al*., 2014). An investigation into the effects of clam dredging in Langstone Harbour, where a modified oyster dredge was used, reported a clear disturbance of sediment (muddy gravel) down to a depth of 15 to 20 cm (EMU, 1992) (see Figure 4 and Annex 7 for example of potential bottom towed gear scars in Langstone Harbour). In southern Portugal, passage of a clam dredge produced a depression 30 cm wide and 10 cm deep (Constantino *et al*., 2009). The presence of dredge tracks may exist for days (Gaspar *et al*., 2003), weeks (Manning and Dunnington, 1995; Mercaldo-Allen & Goldberg, 2011) or months (Wheeler *et al*., 2014). The persistence of dredge tracks may depend on the depth at which they occur. In the Portugal-based study, dredge tracks caused by clam dredging were no longer distinguishable after 24 hours at 6 m depth but remained visible for 13 days at a depth of 18 m (Constantino *et al*., 2009). The magnitude of disturbance is based on the method of harvest, depth of gear penetration (i.e. length of teeth), fishing frequency, towing speed and method of deployment (Mercaldo-Allen & Goldberg, 2011).

Sediment character

Towed demersal fishing gear has been shown to alter sedimentary characteristics and structure, particularly in subtidal muddy sand and mud habitats, as a result of penetration into the sediment (Jones, 1992; Gubbay & Knapman, 1999; Ball *et al*. 2000; Roberts *et al*. 2010). Surface organic material can be mixed into subsurface layers, changing the vertical distribution of sediment layers (Mayer *et al*., 1991; Jones, 1992). Sediment structure may change through the resuspension of sediment, nutrients and contaminants and relocation of stones and boulders (ICES, 1992; Gubbay & Knapman, 1999). Trawling can increase the fraction of fine sediment on superficial layers of the seabed (Queirós *et al*. 2006). As fine material is suspended, it can be washed away from the surface layers (Gubbay & Knapman, 1999). Trimmer *et al*. (2005) reported significant correlations between fishing intensity and sediment silt content (Queirós *et al*. 2006). It is thought that continual sediment resuspension, as a result of trawling, can lead to the accumulation of fine sediments in the superficial layers of sediment in areas that are trawled if there is an absence of significant advective transport (Jennings & Kaiser, 1998; Trimmer *et al*. 2005). Changes in sediment structure from coarse-grained sand or gravel to fine sand and coarse silt has been reported to occur within beam trawl tracks (Leth & Kuijpers, 1996).

Johnson *et al*. (2002) found a number of studies on the effects of otter trawling in gravel and variable habitats and these revealed trawling physically removed fine sediments and biogenic structures through the removal of structure-forming epifauna, moved or overturned stones and boulders, smoothed the seafloor and exposed sediment/shell fragments (Bridger, 1972; Auster *et al*., 1996; Collie *et al*., 1997; Engel & Kvitek, 1998; Freese *et al*., 1999; Johnson *et al*., 2002; Sewell and Hiscock, 2005).

In Estero Bay of the Californian coast, grain size analyses were used to detect any changes in sediment grain size as a result of experimental trawling using a small footrope otter trawl (61 ft head rope, 60 ft ground rope, 8 inch and 4 inch discs, 3.5 ft x 4.5 700 lbs ft trawl doors) (Lindholm *et al*., 2013). The study plots were located at a depth of 160-170 m and sediment analyses revealed the nature of the sediment to be coarse silt/fine sand (Lindholm *et al*. 2013). Post-trawl samples displayed the same grain size distribution as pre-trawl samples, albeit with a slight increase in silt content and 2% decrease in the fine sand fraction (Lindholm *et al*. 2013). Despite these differences, average mean grain size per plot indicated no visible differences between pre- and post- trawl samples and no quantifiable significant sedimentary differences were observed between trawled and control pots or between sample periods (Lindholm *et al*. 2013). These results are supported by a number of other studies including Tuck *et al*. (1998) and Schwinghamer *et al*. (1998), both of which reported no significant differences in sediment grain size in relation to trawling disturbance. Tuck *et al*. (1998) investigated the physical effects of trawling disturbance on a sheltered sealoch in Scotland at 35-40 m depth in an area characterised by 95% silt and clay using modified rockhopper ground gear without a net. Unfortunately, further details on the gear are not available. Schwinghamer *et al*. (1998) examined physical impacts of experimental otter trawling in the Grand Banks in an area of sandy habitat at 120-146 m depth using an Engel 145 otter trawl with 1250 kg oval otter boards and 46 cm rock hopper gear. Despite reporting no change in sediment grain size, acoustic data did reveal that trawling changed small-scale biogenic sediment structures (such as tubes and burrows) down to 4.5 cm (Schwinghamer *et al*. 1998), indicating a reduction in habitat complexity (Løkkeborg, 2005).

Experimental clam dredging activity in Langstone Harbour, using a modified oyster dredge, led to the removal of the coarse fraction of the sediment and larger sand and fine sediment fraction, with minor differences in the silt component (EMU, 1992). The sediment type for this area was muddy gravel (EMU, 1992). In contrast, a study assessing the impacts of suction dredging for common cockle in the Dutch Wadden Sea, revealed a loss of fine silts and subsequent increase in median grain size from 166.2 μm in 1988 to 179.1 μm in 1994 (Piersma *et al*., 2001). The sediment type in the study was sand. In addition, it was speculated that the loss of adult shellfish stocks as a result of suction dredging, may have also resulted in a reduction in the production of faeces and pseudofaeces which contribute to the silt component of the sediment (Piersma *et al*., 2001). The resuspension and dispersal of fine particles can lead to long term effects on particular sieve fraction (Pranovi & Giovanardi, 1994); potentially decreasing the clay portion of the sediment (Maier *et al*., 1998). Other changes in sediment character may also include a lack of consolidation of sediments (Aspden *et al*., 2004), the removal of stones and the removal of taxa that produce structure (i.e. tube-dwelling and burrowing organisms) (Johnson, 2002; Mercaldo-Allen & Goldberg, 2011). Such physical alterations can cause a reduction in sediment heterogeneity and structure available to biota as habitat (Johnson, 2002). In soft sediments, impacts on benthic fauna are likely to change sediment characteristics and vice versa (Piersma *et al*., 2001).

An ongoing study conducted by Leo Clarke at the University of Bournemouth investigated the impacts of clam dredging in Poole Harbour using a BACI (Before-After-Control-Impact) methodology. Core samples were taken from separate areas representing different levels of dredging intensity: an area that has historically been intensively dredged and remains open for a seven month season ('chronic' fishing site); an area that has historically been closed to dredging but will be opened for a five month season ('acute' fishing site); and an area that remains permanently closed to dredging (control site). Interim results indicate a significant effect of site (regardless of time) and of time (regardless of site). Organic content and the volume of fine sediments were found to be highest in the control site and lowest in the chronic fishing site during the study period. Additionally, both organic content and fine sediment volume were observed to decrease in all sites during the study. However, the interaction term between time and site, which would indicate an overall impact of dredging activity in terms of relative change, appears non-significant. While incomplete at the time of writing, the analysis of biological assemblage data indicates that a significant shift in community structure occurred within the acute fishing site during the study period. This shift is characterised by an increase in the abundance of polychaete worm species, but does not constitute a change to the overall biotope composition observed during the study.

Biological disturbance

General ecological issues related to the effects of mechanical shellfish harvesting include resuspension and associated turbidity affects, direct burial and smothering, release of contaminants, release of nutrients, decreased water quality, direct disturbance and removal of infauna and effects on economically important fisheries resources (Coen, 1995). Alterations in particle size and texture may lead to alterations in the type of organisms present in benthic communities (Pranovi and Giovanardi 1994; Skilleter *et al*. 2006). Furthermore, removal of bioturbator species can have indirect ecological effects on the stability and maintenance of biodiversity due to a reduction in habitat complexity (Nilsson & Rosenberg, 2003; Widdicombe *et al*., 2004).

Bottom towed fishing gear has been shown to reduce biomass, production and species richness and diversity (Veale *et al*., 2000; Hiddink *et al*., 2003). Alterations in the size structure of populations and community are also known to occur (Roberts *et al*., 2010). When dredges are towed along the seafloor, surface dwelling organisms can be removed; crushed, buried or exposed and sessile organisms will be removed from the substrate surface (Mercaldo-Allen & Goldberg, 2011). Direct burial or smothering of infaunal and epifaunal organisms is possible due to enhanced sedimentation rates (Mercaldo-Allen & Goldberg, 2011). In a meta-analysis of 39 studies investigating the effects of bottom towed gear, there was an overall reduction of 46% in the abundance of individuals within disturbed (fished) plots (Collie *et al*., 2000). In studies investigating the effect of intertidal dredging, it was common to observe 100% removal of biogenic fauna (Collie *et al*., 2000). This was observed in an experimental study conducted in Langstone Harbour, where the fauna were seen to either be completed removed or considerably reduced by the dredging activity using a modified oyster dredge (EMU, 1992). In the same study, species richness was also found to decrease with a mean number of 6.5 species in the control site compared with 4.4 in the dredge site (EMU, 1992). Another study based in the River Exe in Devon, found that harvesting of manila clams (*Tapes philippinarum*) by hand raking and suction dredging caused an initial reduction of 50% and 90% respectively, in species diversity and abundance (Spencer, 1997). The meta-analysis found that the magnitude of the response of fauna to bottom towed fishing gear varied with gear type, habitat (including sediment type) and among taxa (Collie *et al*., 2000).

In areas that are intensively fished (more than three times per year), the faunal community is likely to be maintained in a permanently altered state and inhabited by fauna adapted to frequent physical disturbance (Collie *et al*., 2000). There is likely to be a shift from communities dominated by relatively high biomass species towards the dominance of high abundances of small-sized organisms (Collie *et al*., 2000). Kaiser *et al*., 2000 reported that regular fishing activity, in the vicinity of the Isle of Man, excluded large-bodied individuals and the resulting benthic community was dominated by smaller bodied organisms more adapted to physical disturbance (Johnson, 2002). The mortality of target and non-target species can also cause an increase in opportunistic species (Wheeler *et al*., 2014). For example, in the initial period after dredging activities, scavenging organisms have been recorded feeding on damaged prey (Gaspar *et al*., 2003).

Whilst dredging causes direct mortality to small and large infaunal and epifaunal organisms, many small benthic organisms such as crustaceans, polychaetes and molluscs, have short generation times and high fecundities, both of which enhance their capacity for rapid recolonization (Coen, 1995). In such instances, the effect of dredging may only be short term. It is thought that short-term and localized depressions in infaunal populations is not a primary concern within subtidal habitats (Coen, 1995).

Moore and Jennings (2000) suggest that fishing with mobile gear has major direct and indirect impacts on seagrass beds. The substrate holding the seagrass beds may be lost or destabilised, seagrass is uprooted and damaged (Tudela, 2004) and re-suspension of sediment reduces light intensity required for photosynthesis (Ardizzone *et al*., 2000).

Numerous studies have shown (Peterson et al 1987, Fonseca et al 1984, Neckles *et al*. 2005 and De Jonge and de Jong 1992) that shellfish dredging immediately reduce shoot density and biomass of seagrass, whilst also increasing the turbidity of the water column (Bishop *et al*., 2005) which in turn has indirect consequences for species assemblages.

Chemical disturbance

The vast majority of experimental studies investigate the physical and biological impacts of demersal trawling & dredging (Johnson *et al*. 2002, Mercaldo-Allen & Goldberg, 2011). Information on the chemical effects of trawling & dredging is therefore very limited (Johnson *et al*. 2002, Mercaldo-Allen & Goldberg, 2011). The chemistry of bottom sediments may be altered when the benthos are disturbed (Mercaldo-Allen & Goldberg, 2011).

Mayer *et al*. (1991) reported the mixing of surface organic material into subsurface layers. This led to the removal of organic matter from the surface metazoan-microbial aerobic chain to an anaerobic system (Jones, 1992). If subsurface layers of sediment are anoxic then further issues may occur and disturbing soft bottom may create anaerobic turbid conditions (Jones, 1992).

A number of studies have reported that sediments become more anoxic after dredging (EMU, 1992; Ferns *et al*., 2000). This may be caused by exposure of deep anaerobic sediment (Johnson, 2002). In one study, a dark anoxic layer was brought to the surface by the action of the harvester on muddy sand, although no such layer presented itself in clean sand (Ferns *et al*., 2000). Disruption of this anoxic layer may result in the release of sulphides into the upper layers of the sediment (Ferns *et al*., 2000). On the other hand, sediments that are overturned by dredging can enhance oxygen penetration into upper sediment layers (Falcão *et al*. 2003).

The removal or disruption to benthic organisms that are involved in biogeochemical processes within the sediment, may alter the biogeochemistry of the sediment (Mercaldo-Allen & Goldberg, 2011). For example, the removal of large benthic bioturbators may affect sediment nutrient and oxygen fluxes ad influence whether the seafloor acts as a source or sink for certain nutrients (Olsgard *et al*., 2008).

3.3.2 Smothering and siltation rate changes; Changes in suspended solids

Smothering effects

The resuspension of fine sediments takes place as fishing gear is towed along the seafloor (Johnson *et al*., 2002). Larger sand particles are redeposited near the dredge whilst measurable amounts of fine silt and clay particles remain in suspension and are potentially transported away by currents (Godcharles, 1971; Tuck *et al*., 2000). The effects of sediment resuspension include increased turbidity and thus a reduction in light, burial of benthic biota, smothering of adjacent areas including potential spawning areas, and negative effects on the feeding and metabolic rates of organisms (Johnson *et al*., 2002). These effects are site-specific and depend on grain size, sediment type, water depth, hydrological conditions, sensitivity of fauna, currents, tides and water mass properties (Coen, 1995).

Resultant sediment plumes and areas of elevated turbidity can extend up to 30 metres beyond the dredge zone (Manning, 1957; Haven, 1979; Manzi *et al*., 1985; Maier *et al*., 1998), potentially transporting and redistributing sediment into adjacent areas (Vining, 1978). In most cases however, the amount of suspended sediment rapidly returns to low levels with distance from the dredge activity (Kyte *et al*., 1976; Maier *et al*., 1998) with 98% resettling within 15 m (Mercaldo-Allen & Goldberg, 2011). Effects of sediment plumes and enhanced turbidity levels appear to be temporary, with the majority of sediment plumes disappearing within hours of dredging (Maier *et al*., 1998). Dispersed sediments may take 30 minutes to 24 hours to resettle (Lambert & Goudreau 1996; Northeast Region EFHSC 200). Shallow water environments with high silt and clay content are likely to experience larger plumes and greater turbidity (Ruffin 1995; Tarnowski 2006).

In the context of natural disturbance, the resuspension of sediment caused by clam dredging in comparison to long-term wind-induced suspension of sediments, may be relatively minor (Auster & Langton 1999). Natural levels of turbidity, generated as a result of winds and tides, can produce particle loads equal to or exceeding that of dredging disturbance (Tarnowski, 2006). Organisms inhabiting inshore environments are therefore adapted to tolerate the resuspension of sediment at a certain level (Tarnowski, 2006). In addition, shellfish dredging only occurs in discrete areas, so the effects caused by resuspension will occur on a much smaller scale than those caused by natural disturbance (Wilber & Clarke, 2001).

The resuspension of sediment can impact upon benthic communities through smothering, burial and increased turbidity. These effects may extend to organisms living a distance away from the fished area (Kyte & Chew, 1975). If high levels of sediment are resuspended and exposure to such events is regular, impacts may be severe (Mercaldo-Allen & Goldberg, 2011). Increased turbidity can inhibit respiratory and feeding functions of benthic organisms, in addition to causing hypoxia or anoxia (Morgan & Chuenpagdee, 2003). Sediment resuspension can jeopardise the survival of bivalves and fish as a result of clogged gills and inhibition of burrowing activity (Dorsey & Pederson, 1998). Small organisms and immobile species are particularly vulnerable to smothering (Manning, 1957). A redistribution of finer sediment can also hinder the settlement of organisms if shell or cultch material is buried (Tarnowski, 2006). Zostera nolti seagrass beds experience 50% shoot mortality when buried in just 2cm of sediment, and 100% in 8cm (Cabaco et al., 2008). The severity of such impacts are largely determined by sediment type, the level of sediment burden and the tolerance of organisms which is largely related to their biology (i.e. size, relationship to substrate, life history, mobility) (Coen, 1995).

Studies conducted in England and Florida found that the redistribution of sediments caused through dredging activity did not result in the smothering of benthic organisms within the nearby area and impacts were found to be limited to the directly disturbed area of the dredge (Schroeder, 1924; Spencer *et al*., 1998). Estuarine ecosystems, where dredging typically takes place, are high variable environments with elevated and variable suspended sediment loads and the organisms living there are often well adapted to such conditions (Coen, 1995). Such organisms are therefore generally considered tolerant to short-term perturbations in sediment loads (Lutz, 1938; Kyte *et al*., 1975). Laboratory experiments have shown that the majority of estuarine infaunal species are able to survive burial depths of up to 20 cm or more (Coen, 1995). In contrast, epifaunal and non-motile species can suffer high mortality rates after burial (Coen, 1995).

3.3.3 Removal of non-target and target species.

Bottom towed fishing gear can result in the mortality of non-target species through direct physical damage inflicted by the passage of the trawl or indirectly through damage, exposure and subsequent predation (Roberts *et al*. 2010). This can lead to long-term changes in the benthic community structure (Jones, 1992), including decreases in biomass, species richness, production, diversity, evenness (as a result of increased dominance) and alterations to species composition and community structure (Tuck *et al*., 1998; Roberts *et al*. 2010). Disturbance from repeated trawling selects for more tolerant species, with communities becoming dominated by smaller-bodied infaunal species with fast life histories, juvenile stages, mobile species and rapid colonists (Engel & Kvitek, 1998; Gubbay & Knapman, 1999; Kaiser *et al*. 2000; Jennings *et al*. 2001; Kaiser *et al*. 2002). In addition, larger individuals may become depleted more than smaller individuals (Jennings *et al*. 2002).

The impacts of fishing activities on benthic communities varies with gear type, habitat and between taxa (Collie *et al*. 2000; Thrush & Dayton, 2002; Kaiser *et al*. 2006). Reported effects are habitat-specific (Roberts *et al*. 2010). A meta-analysis conducted by Kaiser *et al*. (2006) revealed that soft-sediment, especially muddy sands were vulnerable to fishing impacts, with otter trawling and beam trawling all producing a significant immediate impact on this habitat. A number of studies found no detectable impacts, specifically in relation to different forms of trawling in sand habitats (Van Dolah *et al*., 1991; Kaiser & Spencer, 1996; Kenchington *et al*., 2001; Roberts *et al*., 2010), although this is not true in all cases. Such habitats are likely to be pre-adapted to higher levels of natural disturbance and are characterised by relatively resistant fauna (Kaiser *et al*. 2006).

Otter trawls

The impact of otter trawls on benthic communities varies between studies, notably between sediment types. In a meta-analysis of experimental fishing impact studies, conducted by Kaiser *et al*. (2006), otter trawling was found to have one the least negative impacts, compared to other gear and substrata combinations. The initial impact on benthic communities from otter trawl disturbance on mud was estimated to be -29%, -15% on sand and +3% on gravel (Kaiser *et al*., 2006; Hinz *et al*., 2009).

Direct mortality of different megafaunal taxa groups varied after a single sweep with a commercial otter trawl (dimensions unknown) over shallow (30-40 m) sandy areas and deeper (40-50 m) silty sand areas in the southern North Sea (Bergman & van Santbrink, 2000). In areas of silty sand, direct mortality ranged from 0-52% for bivalves, 7% for gastropods, 0-26% for echinoderms, and 3-23% for crustaceans. In areas of sand, direct mortality ranged from 0-21% for bivalves, 12-16% for echinoderms and 19-30% for crustaceans. Experimental otter trawling (dimensions unknown) on the continental shelf of northwest Australia, in an area presumed to be sand, led to an exponential decline in the mean density of macrobenthos with increasing tow numbers (Moran & Stephenson, 2000; Johnson *et al*. 2002). Density was reduced by approximately 50% after four tows and 15% after a single tow (Moran & Stephenson, 2000; Johnson *et al*. 2002). A trawl with 20 cm disks, separated by 30 to 60 cm spacers was used (Johnson *et al*. 2002). No further information on the trawl used is known. The impacts of otter trawling on benthic communities on a sandy bottom in Grand Banks, Newfoundland were studied over a three-year period (Kenchington *et al*., 2001). Three experimental corridors with adjacent reference corridors were established and experimental corridors were trawled 12 times within 5 days for three years using an Engel 145 otter trawl with 1250 kg otter doors, 60 m door spread and 46 cm rockhopper foot gear. Changes in the benthic community were sampled using an epibenthic sledge. The sled is largely used to sample epifauna and some infauna as the sled penetrates to a depth of 2 to 3 cm. Samples collected using the benthic sled revealed a 24% reduction in average biomass in trawled corridors compared to reference corridors. This decrease was caused by reductions in biomass of sand dollars, brittle stars, soft corals, sea urchins and snow crabs. No significant effects were observed for mollusc species. The mean total abundance per grab sample was 25% lower immediately post trawling in one of the three years and declines were demonstrated for 13 taxa primarily made up of polychaetes, which also declined in biomass (Løkkeborg *et al*., 2005).

Valentine and Lough (1991) investigated the impact of scallop dredging and trawling on sand and gravel habitats using side scan sonar and a submersible on eastern Georges Bank. The study documented the most obvious signs of disturbance on gravel pavement habitats. Unfished gravel areas (as a result of the presence of large boulders) had more biologically diverse communities with an abundance of epifaunal organisms. In fished areas, the attached epifaunal community was limited. Similarly, Collie *et al*. (1997) investigated the effects of multiple methods of bottom towing fishing gear (otter trawl and scallop dredging) on benthic megafaunal communities in gravel habitat on Georges Bank at depths between 47 to 90 metres. No information on the types of otter trawls used were given. Numerical abundance of organisms, biomass and species diversity were all significantly greater at undisturbed sites, whilst evenness was greater at disturbed sites (Collie *et al*., 1997). Disturbed sites are likely to have greater evenness because disturbance of towed gear prevents one species becoming numerically dominant (Collie *et al*., 1997). Small fragile polychaetes, shrimps and brittle stars were absent or less common at disturbed sites. At undisturbed sites epifauna such as tube-dwelling polychaetes, bushy bryozoans and hydroids provide a complex habitat.

Engel and Kvitek (1998) documented differences between lightly (average of 220 trawl hours per year) and heavily (average of 816 trawl hours per year) otter trawled areas with similar bottom types (gravel, coarse sand, medium-fine sand and silt-clay) off central California. The densities and abundance of all invertebrate epifaunal species were higher in the lightly fished area when compared to the heavily fished area, including significant differences in species of sea pens, sea stars, sea anemones and sea slugs. Opportunistic species including oligochaetes, nematodes, ophiuroids were found in greater densities in the heavily fished area in each year of the study (1994-1996), whilst significantly more polychaete species were reported in lightly fished areas and no significant difference in the number of crustaceans between the two areas. The study concluded that high levels of trawling can lead to a decrease in habitat complexity and biodiversity and lead to subsequent increases in opportunistic species.

Thrush *et al*. (1998) assessed the importance of fishing pressure (by collecting samples along a fishing pressure gradient) in accounting for variation in community composition in an area characterised by varied sediment characteristics (from 1 to 48% mud) in Hauraki Gulf in New Zealand at depths between 17 to 35 metres. In this area, a major fin fishery for snapper (*Chrysophrys auratus*) exists. The typical trawl gear used consists of 480 kg doors, ground rope of 140-150 mm diameter rubber bobbins, steel balls, with a total ground rope mass of 240 kg (not including sweeps and bridles). After accounting for differences in environmental conditions, the study reported 15-20% of the variability in the macrofauna community composition was attributed to fishing. Observations following reduction in fishing pressures included increases in the density of echinoderms, long-lived surface-dwelling organisms, total number of species, individuals and species diversity. Decreased fishing pressure led to significant increases large epifaunal densities.

Beam trawls

Repeated experimental trawling (3 times) with a 7000 kg, 12 m beam trawl with tickler chains led to a significant 40-65 % decrease in the density of starfishes, small heart urchins, tube-dwelling polychaete worms and small crustaceans, although other species, namely worm and mollusc species, did not change and a number increased (Bergman *et al*. 1990; Bergman & Hup, 1992). The study was conducted in the North Sea in an area of medium hard sandy sediments at a depth of 30 m. Bergman and van Santbrink (2000) reported similar mortality levels of 5-40% in gastropods, starfish, crustaceans and annelid worms and a 20-65% mortality of bivalves using a 12 m and 4 m beam trawl with ticklers and a 4 m beam with chain matrix over shallow sandy areas and deep silty sand areas in the North Sea. Direct mortality in a number of infaunal species was higher in silty areas than in sandy areas (Bergman & van Santbrink, 2000). The 12 m beam trawl caused the highest annual fishing mortality (Bergman & van Santbrink, 2000). In an area of stable coarse sand and gravel, experimental trawling (10 to 12 passes) with a 3.5 tonne 4 m beam trawl with chain matrix led to a 54% reduction in the number of infaunal species and 40% reduction in individuals, a decrease in slow moving epifauna and an increase in mobile species (Kaiser & Spencer, 1996, Kaiser *et al*., 1996, 1998, 1999). At the scale and intensity of the study, no changes in densities were detected (Kaiser & Spencer, 1996, Kaiser *et al*., 1996, 1998, 1999). The same experimental treatment was applied to an area characterised by mobile sand ribbons and megaribbons, however no differences in the benthic community were detected (Kaiser & Spencer, 1996b, Kaiser *et al*., 1996b, 1998, 1999). A study on the impacts of chronic beam trawling in central regions of the North Sea reported significant decreases in infaunal biomass and production in a region of muddy sand sediment and depth of 55 to 75 m (Silver Pit) in response to trawling intensity (Jennings *et al*. 2001). The effects of trawling disturbance were not significant on epifauna and in another region, characterised by sand with a depth of 40-65 m (The Hills) and smaller range of trawling intensity, a relationship between infaunal biomass and production could not be established (Jennings *et al*., 2001). Another study, also based in the central North Sea, investigated the impacts of experimental beam trawling (using a 4 m beam trawl with a chain matrix) on meiofauna and reported that meiofauna are more resistant to trawling disturbance than macrofauna and have the potential to withstand chronic trawling impacts (Schratzberger *et al*. 2002).

Size of fauna

Many studies have observed a shift in benthic community structure from one dominated by relatively high biomass species to one dominated by a high abundance of small-sized organisms (Collie *et al*., 2000). The predicted change in shallow water communities, as a result of trawling disturbance, is an increase in r-strategists (i.e. polychaetes) and decrease K-strategist (i.e. molluscs and crustaceans) (Jones, 1992). A shift towards small-sized species has the potential to alter benthic productivity as body mass is negatively correlated with individual production to biomass ratio (Jennings *et al*., 2001; Queirós *et al*., 2006). Overall reductions in benthic productivity have been reported in areas where intense bottom trawling takes place (Jennings *et al*., 2001). Increases in the biomass or production of smaller infauna have been found to be small in relation to losses in overall community biomass and production that occurred as a result of the depletion of larger individuals (Jennings *et al*., 2001). Smaller bodied fauna are incapable of utilising resources that become available as larger fauna are removed from the community (Queirós *et al*., 2006). Under such conditions, resources may be redirected to other parts of the system (Queirós *et al*., 2006). In areas of natural disturbance, the dominance of smaller bodied fauna may be a general adaptation to such a dynamic environment and therefore the community may seem relatively unaffected by trawling (Queirós *et al*., 2006).

Populations of larger, longer-lived species are less resilient to fishing impacts than smaller, short-lived species as they are able to compensate for any increases in mortality (Roberts *et al*., 2010). In addition, lighter animals are often pushed aside by the pressure wave in front of the net (Gilkinson *et al*., 1998; Jennings *et al*., 2001). Larger fauna are mainly affected through direct physical contact with the gear and may be removed from the community (Bergman & van Santbrink, 2000; Queirós *et al*., 2006). Bergman and van Santbrink (2000) revealed a size-dependent trend for some species with respect to direct mortality from a 12 and 4 m beam trawl. In areas of silty sediments, individuals of the bivalve species *Chamelea gallina* above 2 cm were more vulnerable with mortalities ranging between 22-26%, compared to smaller specimens (4-7% mortality). The impact caused by contact with the fishing gear is not comparable to natural disturbance, and mortalities in more mobile and dynamic sediments will not necessarily be lower than in stable sediments (Bergman & van Santbrink, 2000). The impacts on densities of small individuals may however be greater if the larger animals in question live deeper in the sediment, in addition to their potentially more efficient escape possibilities (Bergman & Hup, 1992; Gubbay & Knapman, 1999).

Studies have shown that trawling impacts on meiofuna (animals that pass through a 500 µm mesh sieve but are retained in a 63 µm mesh sieve) are relatively limited (Brylinsky *et al*., 1994; Scratzberger *et al*., 2002). Brylinsky *et al*. (1994) reported reductions in the abundance of nematodes after experimental flounder trawling on the intertidal in the Bay of Fundy, although the rate of recovery was rapid following trawling disturbance. Scratzberger *et al*. (2002) reported no short- to medium- term (1-392 days after experimental trawling) impacts on diversity or biomass of meiofauna from experimental fishing with a 4 m beam trawl in muddy sand in the southern North Sea. Mild effects on community structure were reported at one location however these impacts were minor in relation to seasonal change. The authors suggested that meiofauna are more resistant to beam trawling than macrofauna and they have the potential to withstand the effects of chronic trawling. Their resistance to trawling is thought to be related to their small body size as they are resuspended rather than killed, combined with their short generation cycles which allow populations to withstand elevated mortality.

Faunal groups and species responses

The relative impact of bottom towed fishing gear on benthic organisms is species-specific and largely related to their biological characteristics and physical habitat. The vulnerability of an organism is ultimately related to whether or not it is infaunal or epifaunal, mobile or sessile and soft-bodied or hard-shelled (Mercaldo-Allen & Goldberg, 2011). Fragile fauna (i.e. bivalves and sea cucumbers) have been shown to be particularly vulnerable to trawling damage and disturbance and sedentary and slowing moving species can be significantly lower (Kaiser & Spencer, 1996; Gubbay & Knapman, 1999). Mobile groups and infaunal bivalves have shown mixed responses to trawling disturbance, with life history considerations such as habitats requirements and feeding modes likely to play a key role in determining a species response (McConnaughey *et al*., 2000; Johnson *et al*., 2002). In a meta-analysis of experimental fishing impact studies, conducted by Kaiser *et al*. (2006), otter trawling was found to have the greatest impact on suspension feeders in mud habitats, perhaps reflecting the depth of penetration from the otter doors, whilst the response of suspension feeders and deposit feeders to beam trawling was highly variable. The most negative effect on deposit feeders was found in gravel habitats and the most negative effect on suspension feeders was found in sand habitats (Kaiser *et al*., 2006). Suspension feeding bivalves, such as *Corbula gibba*, are largely unable to escape burial of more than 5 cm (Maurer *et al*., 1982) and are also sensitive to high sedimentation rates that may occur following intensive trawling (Howell & Shelton, 1970; Tuck *et al*., 1998). Having said this, larger-sized individuals have been shown to be more resistant to trawling disturbance as they are relatively robust (Bergman & van Santbrink, 2000).

Studies have revealed mixed effects on epifauna (organisms that inhabit the seabed surface). Jennings *et al*., (2001) found that chronic trawling disturbance had no significant effect on epifauna in the North Sea. Similarly, no long term effects on the number of epifaunal species or individuals were detected by Tuck *et al*. (1998), although a number of species-specific changes in density did occur (increase in *Ophiura* sp. and decreases in *Hippoglossoides platessoides*, *Metridium senile* and *Buccinum undatum*). The lack of long term effects detected by Tuck *et al*. (1998) is likely to be compounded by the fact that beam trawl gear used was not equipped with a net, as greater effects on epifauna may be expected. The removal of 7 tonnes of epifaunal was reported by Pitcher *et al*. (2000) during experimental trawling, however no significant changes in the density of epifauna were reported (Thrush & Dayton, 2002). Kenchington *et al*. (2001) investigated the impacts of otter trawling on benthic communities on a sandy bottom in Grand Banks, Newfoundland over a three year period. Changes in the benthic community were sampled using an epibenthic sledge. The sled is largely used to sample epifauna and some infauna as the sled penetrates to a depth of 2 to 3 cm. Samples collected using the benthic sled revealed a 24% reduction in average biomass in trawled corridors compared to reference corridors. Hinz *et al*. (2009) investigated the biological consequences of long-term chronic disturbance caused by the otter trawl *Nephrops norvegicus* (Norway lobster) fishery along a gradient of fishing intensity over a muddy fishing ground in the northeastern Irish Sea. The study reported reductions in epifaunal abundance of 81% from the lowest trawling effort recorded (1.3 times trawled/year) to the highest (18.2 times trawled/year). Over the same range of trawl intensities, epifunal species richness decreased by 18%, while no effect was evident for epibenthic biomass.

Epifaunal biomass at high trawling intensity sites was reported to be dominated by *Asterias rubens*, a possible response to elevated food availability in the form of biota killed or damaged by trawling (Hinz *et al*., 2009). Starfish species can respond rapidly to prey availability (Freeman *et al*., 2001) and are known to be resilient from the damaging impacts of trawls (Hinz *et al*., 2009). Similarly, despite lower diversity, a greater dominance of the sea star, *Asterias amurensis*, was reported in heavily fished areas of the eastern Bering Sea (McConnaughey *et al*., 2000). The overall mean abundance of *A. amurensis* was 58.5 kg/ha in the heavily fished, compared with 53.1 kg/ha in the unfished area. In contrast, Bergman and Hup (1992) reported a 43% reduction in the mean density of *A. rubens* after a single beam trawling. Generally speaking, a number of studies have shown to have adverse impacts on echinoderms, including a 0-26% mortality in silty sand and 12-16% mortality in sand as a result of otter trawling in the North Sea (Bergman & van Santbrink, 2000) and a 24% reduction in total biomass of mega-epibenthic species as a result of otter trawling on a sandy bottom in Grand Banks, owing primarily to reductions in sand dollars, brittle stars, soft corals, sea urchins and snow crabs (Kenchington *et al*., 2001). Trawling caused significant damage only to echinoderms, with the highest probability of damage occurring on the sea urchin (10 percent damage) (Kenchington *et al*., 2001). Large and fragile echinoderms particularly suspectible to trawling, include the sea urchins *Brissopsis lyrifera* and *Echinocardium cordatum* (Ball *et al*., 2000), the latter of which has been reported to have a mortality of 10-40% after the single passage of a 4 m and 12 m beam trawl (higher in silty areas than in sandy areas) (Bergman & van Santbrink, 2000). Jennings *et al*. (2001) reported highly significant reductions in the biomass of burrowing sea urchins in response to a chronic beam trawling in the North Sea.

A meta-analysis by Kaiser *et al*. (2006) showed beam trawling in sand to have a greater individual impact on crustaceans, echinoderms and molluscs when compared with annelids, whilst otter trawling in muddy sand appeared to have a greater impact on crustaceans than annelids and molluscs. The single passage of a 4m and 12 m beam trawl in sand and silty sand led to direct mortalities of up to 22% in small-sized bivalves and crustaceans and in megafaunal species up to 68% for bivalves and 49% for crustaceans (Bergman & van Santbrink, 2000). Bivalves such as *Mya truncata*, *Lutraria lutraria* and *Nucula nitidosa* showed greater densities in samples taken after trawling compared to those taken prior to trawling. By contrast, Tuck *et al*. (1998) reported a decline in *Nucula nitidosa* and *Corbula gibba* in abundance in the trawled area relative to reference area, with the former species being identified as sensitive. Other mollusc species reported to be sensitive to trawling disturbance includes the tellin shells, *Tellina fabula* (Bergman & Hup, 1992). Jennings *et al*. (2001) reported highly significant reductions in the biomass of bivalves in response to a chronic beam trawling in the North Sea. The physical interaction with trawl doors with the sea bed was simulated in a test tank in order to examine physical disturbance and biological damage (Gilkinson *et al*., 1998). During the simulation, bivalves which were buried in the scour path were displaced to the berm and 58-70% of displaced individuals were completely or partially exposed on the surface. Despite this, of the 42 specimens in the scour path, only two showed major damage, despite being displaced. A number of studies have reported limited impacts of molluscs in general as a result of trawling disturbance (Bergman & Hup, 1992; Prena *et al*., 1999).

Experimental fishing manipulations have shown that the impacts of trawling disturbance on annelids are limited, and in some instance may be positive, particularly with respect to polychaetes Experimental flounder trawling on an intertidal silty habitat in the Bay of Fundy revealed no impact on either the composition or abundance of polychaetes, the majority of which are tube dwelling (Brylinsky *et al*., 1994). Whilst the single passage of a 4 m and 12 m beam trawl on sandy and silty sediment led to direct mortalities of 31% for annelids, principally the tubedwelling polychaete *Pectinaria koreni*, the mortality of many other small annelids observed was negligible (Bergman & van Santbrink, 2000). Ball *et al*. (2000) reported a decrease in abundance in most species following experimental trawling with a Nephrops otter trawl, except for most polychaete species which increased in abundance following trawling. These species included small opportunistic species such as such as *Chaetozone setosa* (52%), *Prionospio fallax* (149%) and *Scolelepis tridentate* (457%) or large scavenges such as *Nephtys incisa* (16%). Tuck *et al*. (1998) reported a consistently higher proportion of polychaetes in the treatment areas, with an increase in the abundance of opportunistic polychaete species belonging to the cirratulid famly, *Chaetozone setosa and Caullenella zeflandica*, in response to trawling disturbance. The polychaete, *Pseudopolydora paucibranchiata*, also increased in density, immediately following trawling disturbance (Tuck *et al*., 1998). Other polychaete species however did decline in response to fishing disturbance, including *Scolopolos armiger*, *Nephtys cirrosa* and *Terebellides stroemi* (Tuck *et al*., 1998). *Scolopolos armiger* is thought to be sensitive to burial, whilst *N. cirrosa* and T. *stroemi* are larger bodied and therefore more likely to be adversely affected by trawling disturbance (Tuck *et al*., 1998). Bergman and Hup (1992) found that three-fold trawling had minimal effect on the densities of worm species, except for *Magelona*, *Lanice* and *Spiophanes*, although densities of the former species significantly increased after experimental trawling for larger individuals. Jennings *et al*. (2001; 2002) reported no significant changes in polychaetes in in response to a chronic beam trawling in the North Sea. In contrast to the aforementioned studies, Kaiser *et al*., (1998) studied the effect of beam trawling of megafauna in an area of stable sediments in the north eastern and found a reduction the abundance in the polychaetes *Aphtodita aculeata* and *Nephtys* spp., although these differences were no longer apparent 6 months after trawling.

A number of studies have identified common trends for certain species in response to trawling disturbance. The gastropod *Buccinum undatum* is shown to decline in areas of trawling disturbance (Tuck *et al*., 1998; Kaiser *et al*., 2000), with one study stating the effects of trawling persisted for 6 months into the recovery period (Tuck *et al*., 1998). Similarly, *Echinocarodium cordatum* has been identified as a fragile and highly vulnerable to trawling disturbance (Bergman & Hup, 1992; Bergman & van Santbrink, 2000), showing declines of 40 to 60% in density in one study (Bergman & Hup, 1992). Similar reductions were shown by the polychaete *Lanice conchilega* (Bergman & Hup, 1992), a species of polychaete which is highly incapable of movement in response to disturbance and therefore take a significant period of time to recolonise disturbed habitats (Goss-Custard, 1977). Other species that have been reported to exhibit adverse effects of trawling include the polychaete species *Nephtys* (Kaiser *et al*., 1998; Tuck *et al*., 1998) and *Magelona* (Bergman & Hup, 1992; Kaiser *et al*., 2000) and the emergent soft coral *Alcyonium digitatum* (Kaiser *et al*., 1998; 2000; Depestele *et al*., 2012). By contrast, the brittle star, *Ophiura* sp., has been reported to increase or remain constant in response to trawling disturbance (Tuck *et al*., 1998; Gubbay & Knapman, 1999; Kaiser *et al*., 2000; Callaway *et al*., 2007).

Epifaunal organisms inhabiting the seabed surface are subject to crushing or at risk of being buried, in addition to effects of smothering; whilst infaunal organisms living within sediment may be excavated and exposed (Mercaldo-Allen & Goldberg, 2011). A number of studies have found soft-bodied, deposit feeding crustaceans, polychaetes and ophiuroids to be most affected by dredging activities (Constantino *et al*., 2009). This is supported by a meta-analysis conducted by Collie *et al*. (2000) who predicted a reduction of 93% for anthozoa, malacostraca, ophiuroidea and polychaete after chronic exposure to dredging. Furthermore, a study looking at the effects of mechanical cockle harvesting in intertidal plots of muddy sand and clean sand, found that annelids declined by 74% in intertidal muddy sand and 32% in clean sand; and molluscs declined by 55% in intertidal muddy sand and 45% in clean sand (Ferns *et al*., 2000). Similar results were reported by EMU (1992), who found a distinct reduction in polychaetes, but less distinct difference in bivalves, after dredging had taken place and between dredged and control samples. This corresponds with analysis completed by Collie *et al*. (2000) who reported that bivalves appeared to less sensitive to fishing disturbance than anthozoa, malacostraca, ophiuroidea, holothuroidea, maxillopoda, polychaeta, gastropoda and echinoidea,

A number of studies have highlighted species that are particularly vulnerable to dredging as well as those which appear to be more tolerant. For example, the polychaete *Lanice conchilega* is highly incapable of movement in response to disturbance and therefore takes a significant period of time to recolonise disturbed habitats (Goss-Custard, 1977). Deep burrowing molluscs, such as *Macoma balthica*, also have limited capability to escape. Following suction dredging for the common cockle on intertidal sand, the abundance of *Macoma* declined for 8 years from 1989 to 1996 (Piersma *et al*., 2001). Ferns *et al*. (2000) reported reductions of 30% in the abundance of *Lanica conchilega* in intertidal muddy sand after mechanical cockle harvesting (using a tractor) took place, although abundances of *Macoma balthica* increased. The same study also revealed large reductions of 83% and 52% in the abundance of the polychaete *Pygospio elegans* and *Nephtys hombergii*, respectively (Ferns *et al*., 2000). The former species remained significantly depleted in the area of muddy sand for more than 100 days after harvesting and the latter for more than 50 days (Ferns *et al*., 2000). Other polychaete species also thought to be particularly affected are *Arenicola*, *Scoloplos*, *Heteromastus* and *Glycera* (Collie *et al*., 2000).

The aforementioned 8 year decline in *Macoma* following suction dredging for the common cockle on intertidal sand between 1989 and 1996, was also accompanied by a loss of *Cerastoderma edule* (Piersma *et al*., 2001). Declines of bivalve stocks were caused by a particularly low rate of settlement in fished areas (Piersma *et al*., 2001). It is speculated the reason for a lack of settlement was caused by sediment re-working from suction dredging, in particular the loss of fine-grained sediments which are conducive to bivalve settlement (Piersma *et al*., 2001).

3.3.4 Sampling constraints

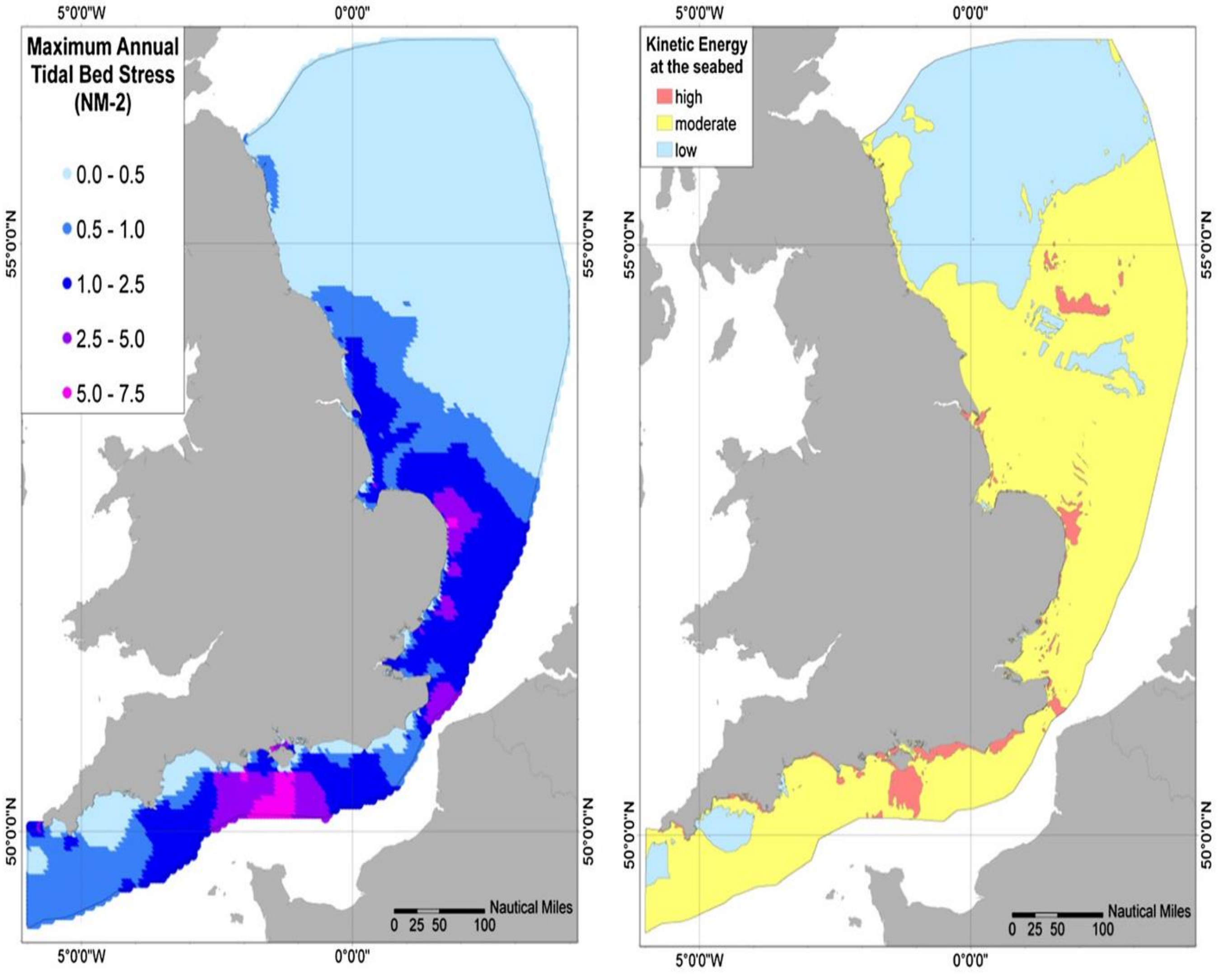
Experimental trawling studies provide a valuable tool for investigating the mechanisms by which bottom-trawl disturbance physically and biologically impacts on benthic habitats (Hinz *et al*., 2009). These experimental fishing manipulations are however often small-scale at spatial scales of km^2 to ha (Hinz *et al*., 2009). Some contain the caveat that the study area chosen may have been markedly affected by previous fishing activities (Tuck *et al*., 1998). If there are substantial changes in the benthic community in the initial period of trawling development, it may be difficult to detect subsequent trends or impacts from fishing because the community is resistant to such effects or because effects are relatively insignificant compared to those caused previously (Tuck *et al*., 1998). The benefits of using pristine, unfished sites which are then subject to experimental trawling gives a good idea of a benthic community's response and allows recovery to be quantified following fishing disturbance (Hinz *et al*., 2009). These findings provide helpful indications of instantaneous effects and relative severity of impacts for different gear types (Collie *et al*., 2000; Kaiser *et al*., 2006). Comparisons of high, low or no fishing intensity involves the classification of such areas in these fishing intensity levels (Hinz *et al*., 2009). These are often relative measures that are specific to each study, limiting generality and comparability (Hinz *et al*., 2009). Study sites chosen as unfished sites are often inaccessible to fisheries due to an obstruction and these can generate confounding effects (Hinz *et al*., 2009). Likewise, areas used as control sites may be subject to different environmental conditions, leading to further confounding effects (Hinz *et al*., 2009).

Experimental studies do however have a number of significant limitations (Hinz *et al*., 2009). Quantifying the effects of fishing impacts under realistic fishing conditions is difficult and the spatial and temporal scale of disturbance generated by a trawling fleet is unfeasible in an experimental context (Hinz *et al*., 2009). The occurrence of chronic fishing disturbance over large spatial scales can be expected to lead to greater effects and slower recovery rates than those reported in experimental studies (Hinz *et al*., 2009).

3.3.5 Natural disturbance

Communities that exist in areas of high natural disturbance rates are likely to have characteristics that provide resilience to additional disturbance (Hiddink *et al*., 2006a). Any vulnerable species would be unable to exist within conditions of frequent disturbance (Hiddink *et al*., 2006a). The impact of trawling is therefore expected to be higher in areas that experience low levels of natural disturbance and lower at locations of high levels of natural disturbance (Hiddink *et al*., 2006a). Despite the significance between benthic community responses to trawling disturbance and levels of natural disturbance, the relationship remains unquantified (Hiddink *et al*., 2006a). There can often be a failure to detect the effect of experimental fishing disturbance in areas exposed to high levels of natural disturbance (Thrush & Dayton, 2002). Whilst it may be appropriate to equate effects of natural disturbance to some effects of trawling disturbance, it is not always the case. Fishing can involve a higher intensity of disturbance, although this is dependent on frequency and extent (Thrush & Dayton, 2002). A trawl effects small-sized organism through sediment perturbations, which is comparable to that of natural disturbance, whereas its impacts on larger-bodied organisms will be through physical contact with fishing gear (Bergman & van Santbrink, 2000). The relatively low impact on benthic communities inhabiting mobile sediments might therefore only apply to small-bodied animals (Bergman & van Santbrink, 2000).

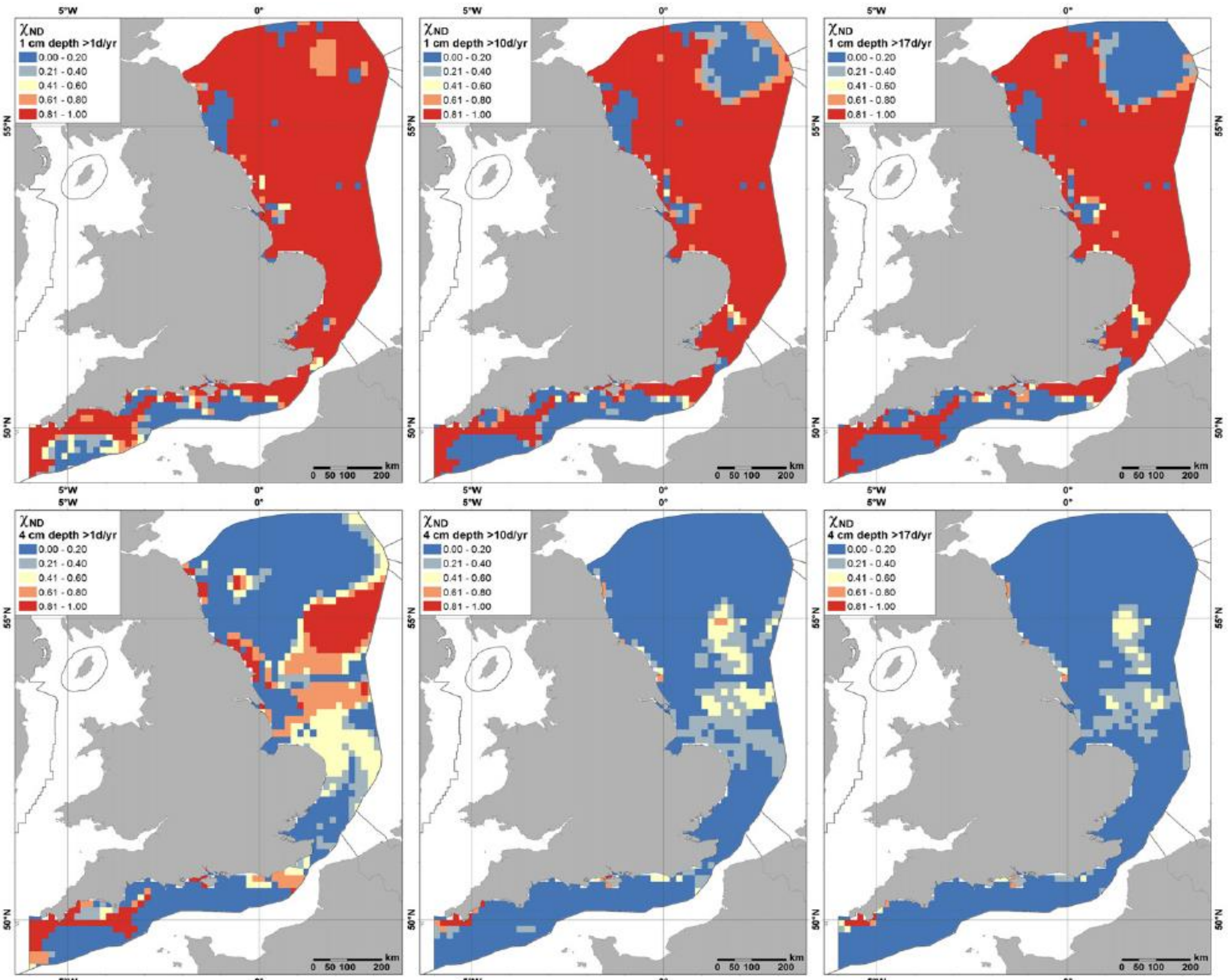
Figure 5. Maps of modelled natural disturbance of the seabed, represented by tidal bed stress (left) and kinetic energy (right). Source: Bolam et al., 2014.



Measures used to detect changes in the benthic community (i.e. abundance, biomass) can be subject to considerable temporal variability and make it difficult to detect any changes caused by trawling disturbance (Løkkeborg, 2005). A number of studies have shown that control areas experience considerable change throughout the duration of a study and such temporal changes occur irrespective of trawling disturbance (Kenchington *et al*., 2001; Løkkeborg, 2005). It can be difficult to attribute long-term changes to benthos to trawling alone, since other forces are likely to be acting on the community, including natural fluctuations, chemical dumping and eutrophication (Pearson & Barnett 1987; Rees & Eleftheriou 1989; Jones 1992). Sanchez *et al*. (2000) concluded the decrease in certain species in unfished areas was likely to indicate natural variability at the site exceeds the effects of fishing disturbance. Similarly, Kaiser *et al*. (1998) concluded that only subtle changes in community structure were caused by trawling and effects caused by seasonal fluctuations and natural disturbance were more pronounced (Løkkeborg, 2005).

Studland and Poole Bay is relatively sheltered and therefore may not experience high levels of natural disturbance. Bolam *et al*. (2014) modelled natural seabed disturbance as part of a study looking at the sensitivity of microbenthic second production to trawling in the English sector of the greater North Sea. Natural seabed disturbance was represented by tidal bed stress and kinetic energy at the seabed. Maps showing the probability of natural forces disturbing the seabed to 1 and 4 cm for a range of frequencies (once, 10 times, and 17 times were also created. These maps cover Studland (Figures 5 and 6), although the resolution is low as the area covered includes the North Sea and western English Channel. The maps demonstrate that the areas is exposed to low to medium levels of natural disturbance. Annual tidal bed stress ranges between 0.0 and 0.5 NM^2 and kinetic energy at the seabed is moderate. The probability of natural forces disturbing the seabed to 1 cm are at lowest probability (0.00 – 0.20) at all frequencies.

In the context of MPA management, it is important to qualify which changes occur to naturally dynamic communities as a result of natural variability within the environment, as opposed to that resulting from anthropogenic pressures (Goodchild *et al*., 2015). The reason being that the conservation objectives of a site



are 'subject to natural change (Goodchild *et al*., 2015). It can therefore prove difficult in ascertaining if the conservation objective of a site is being compromised by anthropogenic pressures if the MPA feature is also subject to natural variability (Goodchild *et al*., 2015). Potential changes caused by towed fishing gear could be masked by the impacts of natural sediment movements which maintain the benthic community in a state of successional flux (Løkkeborg, 2005; Goodchild *et al*., 2015). A recent study attempted to analyse existing data to study effects of towed fishing gears on mobile sediments against a background of natural variability, however, it concluded the results of the study were of little direct value in terms of MPA management (Goodchild *et al*., 2015).

Figure 6. Maps of the modelled probability that natural forces disturb the seabed to different depths of 1 and 4 cm for a range of frequencies per year (once, 10 and 17 times). Source: Bolam et al., 2014

3.3.6 Sensitivity

Habitat type

In a meta-analysis of 39 studies, which were conducted on varying sediment types, the most negative impacts occurred in muddy sand and gravel habitats (Collie *et al*., 2000). Surprisingly, the meta-analysis revealed the least impact was observed on mud habitats and not sand, which was not consistent for the results obtained for abundance and species richness (Collie *et al*., 2000). It was however noted that this may have been explained by the fact most studies conducted on mud habitats were looking at the impacts of otter trawls and that if data were available for the effect of dredgers a more negative response for this habitat may have been observed (Collie *et al*., 2000). In a separate meta-analysis of 101 different fishing impact manipulations, the initial and long term impacts of different fishing types were shown to be strongly habitat-specific (Kaiser *et al*., 2006). Gravel habitats were negatively affected in both the short and long term by scallop dredging whilst soft-sediments (especially muddy sand) were particularly vulnerable to fishing impacts, with intertidal dredging shown to have the most severe initial impact (Kaiser *et al*., 2006; Roberts *et al*., 2010).

Moschino *et al*. (2003) reported enhanced damage to the clam *Chamelea gallina* in fine grain sand compared to those on coarser sand as a result of experimental hydraulic dredging. Another study by Ferns *et al*. (2000) observed a quicker recovery of species in an area of intertidal sand compared with an area of intertidal muddy sand. Recovery of individual species population densities in intertidal sand were reported to take up to 39 days, compared with over 174 days for some species in intertidal muddy sand (Ferns *et al*., 2000).

Beam trawling had significant negative short-term impacts in sand and muddy sand habitats, although the relative effect was less and recovery times shorter than for intertidal dredging (Kaiser *et al*., 2006). Otter trawling had a significant initial effect on muddy sand and mud habitats, although long-term impacts, post trawling, on mud habitats were positive (Kaiser *et al*., 2006). The initial impact on benthic communities from otter trawl disturbance on mud was estimated to be -29%, -15% on sand and +3% on gravel (Kaiser *et al*., 2006; Hinz *et al*., 2009).

A number of studies have found limited detectable impacts of trawling in sand habitats (Van Dolah *et al*., 1991; Kaiser & Spencer, 1996; Kenchington *et al*., 2001; Roberts *et al*., 2010). Queirós *et al*. (2006) investigated the impact of chronic trawling on two communities from a muddy sand and a sandy habitat in the Irish Sea and Dogger Bank respectively. Chronic trawling was found to have an adverse effect on the biomass and production of benthic communities in muddy sand, whilst no impact was identified on benthic communities within the sandy habitat. It is important to note the two areas are fished with different gear types; the Dogger Bank is mostly fished by beam trawlers targeting plaice and the Irish Sea is fished by otter trawls targeting Norway lobster. Another study by Lindholm *et al*. (2013) reported no measurable effects of otter trawling using a small footrope otter trawl on the density of benthic invertebrates in areas of coarse silt/fine sand.

Bolam *et al*. (2014) investigated the relative sensitivity of benthic macrofauna to trawling, both short- and long-term and used this information to describe the spatial variation in sensitivity of secondary production. In general, it was found that the more sensitive and productive regions (northern North Sea and western English Channel) are associated with poorly-sorted, gravelly or muddy sediments, whilst less sensitive and less productive regions (southern North Sea) are associated with well-sorted sandy sediments (Bolam *et al*., 2014). Faunal assemblages, whose total production has a low overall sensitivity to trawling, occur in sandy sediment sediments containing low silt/clay and/or gravel fractions and such sensitivity inversely correlates with levels of natural disturbance. Thus, total production is more sensitive to trawling in deep regions with little or no natural sediment disturbance (Bolam *et al*., 2014). This is largely driven by long-term sensitivity of taxa and less so by instantaneous sensitivity (Bolam *et al*., 2014).

The reason for the sensitivity of different sediment types to the impacts of bottom towed fishing gear is related to the physical stability of the seabed (Collie *et al*., 2000). Fauna living within unconsolidated sediments such as those in shallow and sandy environments, are more adapted to dynamic environments, periodic resuspension and smothering and therefore able to recover more quickly (Tuck *et al*., 1998; Collie *et al*., 2000). Experimental studies investigating disturbance in shallow sandy environments indicate changes in community response are generally short-term (Kaiser *et al*., 1998) or non-existent (Queirós *et al*., 2006; Lindholm *et al*., 2013). Impacts of bottom towed gear are therefore greatest in areas with low levels of natural disturbance (Hiddink *et al*., 2003).

Sensitivity analyses

A number of recent studies have endeavoured to map the sensitivity of habitats to different pressures (Tillin *et al*., 2010) and fishing activities (Hall *et al*., 2008).

Tilin *et al*. (2010) developed a pressure-feature sensitivity matrix, which in effect is a risk assessment of the compatibility of specific pressure levels and different features of marine protected areas. The approach used considered the resistance (tolerance) and resilience (recovery) of a feature in order to assess its sensitivity to relevant pressures (Tilin *et al*., 2010). Where features have been identified as moderately or highly sensitive to benchmark pressure levels, management measures may be needed to support achievement of conservation objectives in situations where activities are likely to exert comparable levels of pressure (Tilin *et al*., 2010). In the context of this assessment, the relevant pressures likely to be exerted are penetration and abrasion of the seabed and removal of non-target species. Sensitivity of subtidal sediment types to these pressures vary from not sensitive to high, generally with low confidence in these assessments (Table 7). Subtidal mixed sediments appear to be sensitive overall, followed by subtidal mud, whilst subtidal coarse sediment and sand appears to has relatively low sensitivity overall.

Hall *et al*. 2008 aimed to assess the sensitivity of benthic habitats to fishing activities. A matrix approach was used, composed of fishing activities and marine habitat types and for each fishing activity sensitivity was scored for four levels of activity (Hall *et al*., 2008). The matrix was completed using a mixture of scientific literature and expert judgement (Hall *et al*., 2008). The type of fishing activities chosen were ‘beam trawl & scallop dredges’ and ‘demersal trawls’ as these encompassed the fishing activities under consideration. Generally, stable habitat types exhibit high sensitivity to heavy gear intensities for beam trawls and scallop dredges and demersal trawls (Table 8). A large number of habitat types exhibit medium sensitivity to moderate gear intensities, except for beam trawls and scallop dredges in subtidal muddy sand and stable rich mixed sediments. All habitat types, except stable rich mixed sediments, exhibit low sensitivity to light fishing intensity and all habitat types exhibit low sensitivity to a single pass (Table 8). Generally, sensitivity across all habitat types is lower for light demersal trawls and seines, as would be expected (Table 8).

Table 4. Sensitivity of SAC features to pressures identified by Tillin et al. (2010). Confidence of sensitivity assessment is included in brackets.

	Pressure					
Feature	Penetration and/or disturbance of the substrate below the	Shallow abrasion/penetration – damage to seabed surface and penetration <25mm	Surface abrasion: damage to seabed	Removal of non-target species	Removal of target species	Siltation rate changes (low)

	surface of the seabed – structural damage to seabed >25mm		surface features			
Subtidal sand	Low – Medium (Low to Medium)	Not Sensitive - Medium (Low)	Not Sensitive – Medium (Low)	Not Sensitive – Medium (High)	Not sensitive (low)	Medium (low)
Seagrass Beds	High (low)	High (High)	Low (Low)	High (high)	Not Sensitive (high)	High (medium)

Table 5. Sensitivity of SAC features to different intensities (high, medium, low, single pass) of oyster/mussel dredging as identified by Hall et al. (2008).

Gear Type	**Habitat Type**	**Gear Intensity***			
		Heavy	Moderate	Light	Single pass
Beam trawls & scallop dredges	**Subtidal stable muddy sands, sandy muds and muds**	High	High	Low	Low
	Stable subtidal fine sands	High	Medium	Low	Low
	Seagrass beds	High	High	High	High
Demersal trawls	**Subtidal stable muddy sands, sandy muds and muds**	High	Medium	Low	Low
	Stable subtidal fine sands	Medium	Medium	Low	Low
	Seagrass beds	High	High	High	High
Light demersal trawls and seines	**Subtidal stable muddy sands, sandy muds and muds**	Medium	Low	Low	Low
	Stable subtidal fine sands	Medium	Medium	Low	Low
	Seagrass beds	High	High	High	High

*Gear activity levels are defined as follows; Heavy – Daily in 2.5 nm x 2.5 nm, Moderate – 1 to 2 times a week in 2.5 nm x 2.5 nm Light – 1 to 2 times a month during a season in 2.5 nm x 2.5 nm, Single pass – Single pass of fishing activity in a year overall

3.3.7 Recovery

Recovery ultimately depends on the level of impact which is related to the weight of gear on the seabed, towing speed, the nature of bottom sediments and strength of tides and currents (Jones, 1992).

Habitat type and biological recovery

The timescale for recovery largely depends on sediment type, associated fauna and rate of natural disturbance (Roberts *et al*., 2010). Experimental studies have reported a variety of responses to trawling disturbance (Dernie *et al*., 2003). Such variation arises from characteristics specific to the site, i.e. location, gear fishing, season and habitat (Dernie *et al*., 2003). This hinders the formation of general conclusions and recovery rates of communities that would of use for ecosystem management (Dernie *et al*., 2003).

Generally speaking, in locations where natural disturbance levels are high, the associated fauna are characterised by species adapted to withstand and recover from disturbance (Collie *et al*., 2000; Dernie *et* *al*., 2003; Roberts *et al*., 2010). More stable habitats, which are often distinguished by high diversity and epifauna, are likely to take a greater time to recover (Roberts *et al*., 2010). In a relatively recent meta-analysis on the biological impacts of different fishing activities, recovery of muddy sands was predicted to take months to years and sand was predicted to take days to months (Kaiser *et al*., 2006). Similarly, Dernie *et al*. (2003) reported clean sand communities to have the most rapid rate of recovery following disturbance, with muds having an 'intermediate' recovery rate and muddy sand habitats having the longest recovery rates. More specifically, Kaiser *et al*. (2006) reported recovery times in the abundance of biota of less than 50 days from beam trawling in highly energetic, shallow, soft-sediment habitats of sand and muddy sand. In more stable gravel sediments, biota were still reduced by 40% after 50 days (Kaiser *et al*., 2006). Collie *et al*. (2000) reported recovery times of 100 days in sandy sediment communities from trawling disturbance. Kaiser *et al*. (1998) investigated the impacts of beam trawling on megafaunal communities in two areas characterised by mobile megaripple structures and stable uniform sediments. Effects of trawling in mobile sediments were not detectable and in uniform sediments were no longer evident after 6 months (Kaiser *et al*., 1998). The impacts of otter trawling on benthic communities on a sandy bottom in Grand Banks, Newfoundland a 120-146 m depth was studied over a three-year period (Kenchington *et al*., 2001). The sampling programme was not designed to determine the long-term effects and recovery, although available data indicated a recovery of the habitat and biological community within a year or less (Løkkeborg, 2005). Tuck *et al*. (1998) studied the biological effects of otter trawling in a sheltered sealoch in Scotland at 35-40 m depth in an area characterised by 95% silt and clay. A similar condition to the reference site was reached after 18 months, with the abundance of individuals shown to return to similar levels recorded prior to trawling (Tuck *et al*., 1998). Partial recovery of infaunal species occurred after 12 months and effects on epifauna were largely indistinguishable from the reference site 6 months after fishing ceased (Tuck *et al*., 1998; Johnson *et al*., 2002). Brylinsky *et al*. (1994) reported a rapid recovery of nematode abundance within 4 to 6 weeks following experimental flounder trawling on intertidal silty sediments in the Bay of Fundy.

Foden *et al*. (2010) investigated recovery of different sediment types based on the spatial and temporal distribution of benthic fishing. Vessel monitoring system data (2006 to 2007) was used to estimate the distribution and intensity of scallop dredging, beam trawling and otter trawling in UK marine waters. This data was then linked to habitat in a geographic information system. Recovery periods for different habitats were estimated based on existing scientific literature for gear types and fishing intensity (Table 10), with recovery rates generally increasing with sediment hardness. It was estimated that based on mean annual trawl frequencies that 80% of bottom-fished areas were able to recover completely before repeat trawling. In 19% percentage bottom-fished areas however, the frequency of scallop dredging in sand and gravel and otter trawling in muddy sand and reef habitats occurred at frequencies that prevented full habitat recovery. At average fishing intensities (for each gear type), sand and mud habitats were able to recover fully, whilst gravel, muddy sand and reef habitats were fished at frequencies in excess of the estimated recovery period (shown in Figure 6 where the mean index of recovery exceeds 1).

Gear Type	**Habitat Type**	
	Sand	Muddy sand
Beam trawl	182[a]	236[b]
Otter trawl	0[b]	213[c]
Scallop dredge	2922[b,d]	589[b]
[a] Kaiser *et al*. (1998); [b] Kaiser *et al*. (2006); [c] Ragnarsson & Lindegarth (2009); [d] Gilkinson *et al*. (2005)		

Table 6. Recovery rates (days) of different habitats for different fishing gear types. ND: No Data. Source: Foden et al., 2010.

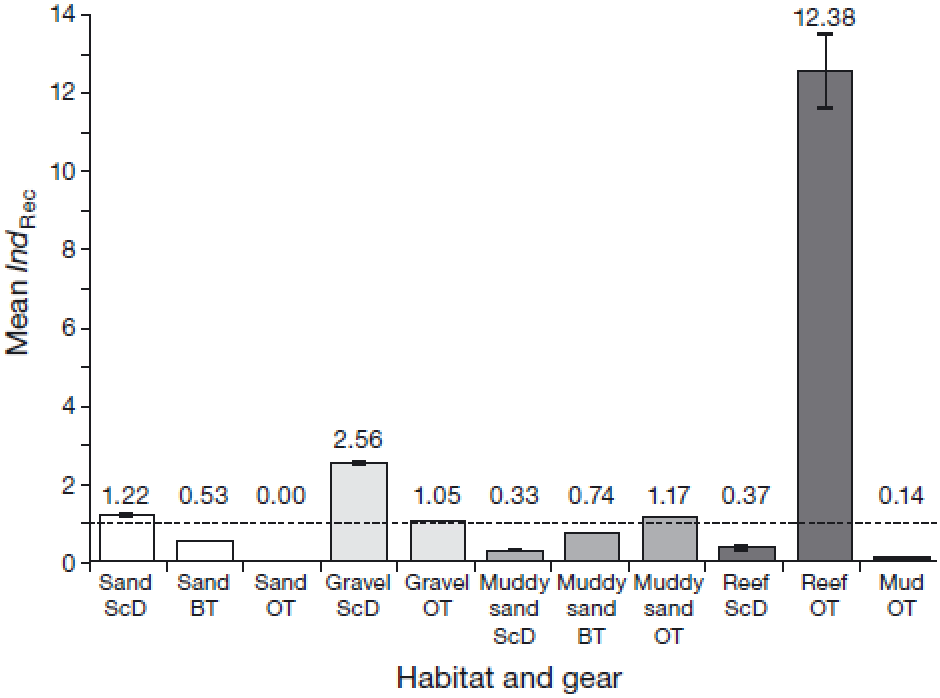


Figure 7. Mean index of recovery (Ind_{Rec}) for gear-habitat combinations using fishing intensity data derived from Vessel Monitoring Systems in 2007. At Ind_{Rec} Rec = 1, the recovery period is equal to fishing frequency (horizontal dashed line), at Ind_{Rec} <1 fishing frequency is less than the predicted recovery period and at Ind_{Rec} fishing frequency exceeds the recoveyr period. BT: Beam Trawl, OT: Otter Trawl and ScD: Scallop Dredge. Source: Foden et al., 2010.

Physical disturbance from chronic trawling occurs over large spatial scales and it may be expected that recovery rates will be slower than those assumed from experimental studies (Hinz *et al*., 2009). Recovery at small experimental scales is likely to simply be immigration, which is a form of recovery that is unlikely in large and repeatedly trawled areas (Jennings *et al*., 2001). The recovery of chronically disturbed benthic communities on fishing grounds will be largely dependent on recruitment and population growth, rather than on immigration from adjacent untrawled areas (Hiddink *et al*., 2006b). The importance of larval recruitment for the recolonization of a disturbed area increases with the size of the disturbed area (Smith & Brumsickle, 1989; Foden *et al*., 2010). The time of year when disturbance takes place may also influence the mode of recovery and recovery rate of the affected community (Foden *et al*., 2010). The recruitment supply of larvae and adult infauna will vary at different times of year and in relation to the physical characteristics at a specific location (Foden *et al*., 2010). The hydrodynamic regime will influence the rate of recolonization by influencing the deposition of infaunal adults and larval stages (Foden *et al*., 2010).

The recovery periods for sandy habitats is estimated to take days to months (Kaiser *et al*., 2006). In the meta-analysis conducted by Kaiser *et al*. (2006), a significant linear regression with time for the response of annelids to the impacts of intertidal dredging in sand and muddy sand habitats was reported. Annelids were predicted to have recovered after 98 days post fishing in sand habitats and 1210 days in muddy sand habitats (Kaiser *et al*., 2006). Authors stated recovery for the latter however should be treated with caution (Kaiser *et al*., 2006).

The longer recovery periods for soft sediments are related to the fact these habitats are mediated by physical, chemical and biological processes, as opposed to the dominance of physical processes that occur within sandy habitats (Roberts *et al*., 2010). Furthermore, the recolonization of soft sediment habitats requires the recruitment of larvae, compared with migration of adult organisms in sandy habitats (Kaiser *et al*., 2006).

Population recovery rates are known to be species specific (Roberts *et al*., 2010). Long-lived bivalves will undoubtedly take longer to recovery from disturbance than other species (Roberts *et al*., 2010). Megafaunal species such as molluscs and shrimp over 10 mm in size, especially sessile species, are more vulnerable to impacts of fishing gear than macrofaunal species as a result of their slower growth and therefore are likely to have long recovery periods (Roberts *et al*., 2010). Short-lived and small benthic organisms on the other hand have rapid generation times, high fecundities and therefore excellent recolonization capacities (Coen, 1995). For example, slow-growing large biomass biota such as sponges and soft corals are estimated to take up to 8 years, whilst biota with short life-spans such as polychaetes are estimated to take less than a year (Kaiser *et al*., 2006).

Habitat type and physical recovery

Like the biological recovery of faunal communities, the physical recovery of sediments is largely related to sediment types and can be very site-specific (Mercaldo-Allen & Goldberg, 2011). In high energy environments physical recovery can take days, whereas recovery in low energy areas can take months (Northeast Region EFHSC, 2002; Wallace & Hoff, 2005). Dredge tracks persist for longer periods of time when there is less energy to erode dredge tracks (Mercaldo-Allen & Goldberg, 2011). The dredge associated trenches have found to be deeper and persistent for longer periods on sandy-mud habitats when compared with sand (Gaspar *et al*., 2003). Dredge tracks sandy and coarse sediment habitats are relatively short-lived and can disappear within 24 hours (Gaspar *et al*., 1998; 2003), although can last a few days to no more than a year (De Groot & Lindeboom, 1994; Lindeboom & de Groot, 1998). This is a relatively short period of time and dredge tracks have been known to persist on timescales from days to weeks to months (Gaspar *et al*., 2003; Manning & Dunnington, 1955; Mercaldo-Allen & Goldberg, 2011). Using side scan sonar and underwater video technology, Smith *et al*. (2007) showed trawl impacts on silty clay sediment were evident through the year within the study area, which also included a closed season. Marks left by a hydraulic dredge at a site in England were no longer obvious after 11 weeks (Tuck *et al*., 2000), although it took seven months to restore sediment structure after suction dredging at a separate site in England (Kaiser *et al*., 1996).

Marks left by dredging may no longer be visible after a certain period of time but differences in sediment composition may still be detectable (Mercaldo-Allen & Goldberg, 2011). Using acoustic reflective sonar, long-term changes in sediment structure has been detected between dredge furrows and the surrounding seabed (Mercaldo-Allen & Goldberg, 2011). One year after the use of an escalator harvester in Maryland, the substrate exhibited less compaction, increased porosity and softer substrates (Pfitzenmeyer, 1972a; 1972b). In Florida, differences in sediment composition between dredged and undredged areas after hydraulic escalator harvesting were no longer present after 1 year (Godcharles, 1971).

The persistence of dredge scars does not necessarily indicate a lack of biological recovery. Dredge scars are likely to persist in areas characterised by low energy, during which time biological recovery may have taken place. It is therefore important to consider the type of environment in which the scars are present as biological recovery may take place over shorter timescales.

The persistence of marks produced as a result of trawling depend on a number of factors including their depth, sediment type, current, wave action and biological activity (Tuck *et al*., 1998; Fonteyne, 2000; Smith *et al*., 2000; Humborstad *et al*., 2004 in Løkkeborg *et al*., 2005). In high energy environments physical recovery can take days, whereas recovery in low energy areas can take months (Northeast Region EFHSC, 2002; Wallace & Hoff, 2005). Trawl marks persist for longer periods of time when there is less energy to erode these marks (Mercaldo-Allen & Goldberg, 2011). Marks are likely to persist longer in deep water and in sheltered areas with fine sediments (Tuck *et al*., 1998; Løkkeborg *et al*., 2005). Trawl marks in areas of faster water movement are likely to be filled in within a shorter period (Jones, 1992).

Marks from towed gear have been showed to be relatively short lived in coarse sediments, lasting from a few days to no more than a year (De Groot and Lindeboom, 1994; Lindeboom & de Groot 1998). In a sandy habitat on the Grand Banks at 120-146 m depth, marks left by trawl doors (1250 kg oval otter boards) were visible for at least 10 weeks, although were not visible or faintly visible after a year (Schwinghamer *et al*. 1998). Tracks from a 4 metre beam trawl with tickler chain matrix remained visible for 52 hours in coarse sand and 37 in fine sand at a depth of 20 to 30 metres on the Goote Bank off Belgium and the Netherlands (Fonteyne, 2000). Trawl door scars (10 cm deep and 20 cm wide) from 2300 kg trawl doors on a sandy/gravel bottom were shown to disappear within less than five months in an area of strong currents in the Barents Sea (Humborstad *et al*. 2004). Hand-dug trenches (15 cm deep and 1.2 m long) at a 7 m deep sandy site lasted for 1 to 4 days in Narragansett Bay, Rhode Island (DeAlteris *et al*., 1999). In the same study, but in the areas of mud at a depth of 14 m, trawl scars (5-10 cm deep with berms 10-20 cm high) persisted for more than 60 days (DeAlteris *et al*. 1999).

In areas characterised by silt or mud, tracks and scars appear to remain visible for longer periods of time compared to sandy and coarser sediments as expected. In a sheltered sealoch in Scotland characterised by sediment with 95% silt and clay, side-scan results revealed that disturbance tracks could still be seen after 18 months after experimental trawling had ceased (Tuck *et al*., 1998). An alternative measure of seabed properties were altered by fishing was also obtained from RoxAnn measurements (Tuck *et al*. 1998), an acoustic bottom classification system based on the seabeds hardness and roughness (Løkkeborg, 2005). RoxAnn data however indicated recovery after 6 month for physical effects (Tuck *et al*. 1998). Smith *et al*. (2007) also used side scan sonar, as well as underwater video technology, to record the impact of trawling on silty clay sediment at depths of 200 m in Herkalion Bay (Roberts *et al*., 2010). Trawl marks were evident throughout the year in the study area, including throughout a closed season of four months, by the end of which trawl marks were less visible indicating biogenical weathering (Smith *et al*. 2007; Roberts *et al*., 2010). No information on the gear type was given. Furrows (5 cm deep, 30-85 cm wide) made by experimental flounder trawl doors (200 kg) in the Bay of Fundy were visible for at least 2 to 7 months in an area of coarse sediment overlain by up to 10 cm of silty sediment (Brylinsky *et al*. 1994).

The persistence of trawl scars does not necessarily indicate a lack of biological recovery. Trawl scars are likely to persist in areas characterised by low energy, during which time biological recovery may have taken place. It is therefore important to consider the type of environment in which the scars are present as biological recovery may take place over shorter timescales.

Studies on recovery rate

There are a limited number of studies which examine the recovery rate from biological and physical disturbance caused by shellfish dredging. Five studies were found on the impacts of shellfish harvesting on intertidal habitats, four of which are based in the UK (details are provided in Annex 9). The recovery rates reported range from no effect (thus no recovery is required) up to 12 months, with intermediate recovery rates reported at 56 days and 7 months (Kaiser *et al*., 1996; Hall & Harding, 1997). Spencer *et al*. (1998) reported a recovery rate of up to 12 months, although inferred it was not possible to be certain recovery had not occurred before this as not all treatment replicates were taken 4 and 8 months after sampling. The authors compared their findings with similar studies and speculated the greater length of recovery in comparison was related to the protected nature of the site (Spencer *et al*. 1998). This study highlights the importance of exposure in determining recovery rates of different habitats and also how recovery rates are site-specific.

Ferns *et al*. (2000) examined the recovery rates of individual species and found the rate of recovery varied between sediment types (muddy sand versus clean sand).

The recovery of seagrass beds are highly variable and are dependent on the extent of removal. Rates may be slow where adjacent seed sources and viable grass beds are present, but can take between 60 and 100 years where the removal of rhizomes has occurred (Gonzalez-Correa *et al*., 2004; Moore and Jennings, 2000).

Depth

There is an inverse relationship between wave action and depth and so the natural mobility of bottom sediments tends to decrease with depth (Wheeler *et al*., 2014). The impact of trawling and dredging might therefore be more substantial in deeper subtidal habitats due a lack of water movement (Jones, 1992, Wheeler *et al*., 2014).

In a literature review by Johnson *et al*. (2002), studies which took place at greater depths (>120 m) revealed trawling tracks were evident up to a year after trawling, whilst those at shallow sites (<7m) were no longer visible after a few days.

Benthic communities in dynamic shallow water are likely to be more capable of overcoming disturbance than those in inhabiting deeper and less dynamic environments and as such are likely to have longer recovery times (Jones, 1992).

3.4 Existing management measures

- **Vessel Used in Fishing** byelaw – prohibits commercial fishing vessels over 12 metres from the Southern IFCA district. The reduction in vessel size also restricts the type of gear that can be used, with vessels often using lighter towed gear and restricted to carry less static gear.
- Southern IFCA has a **Minimum Fish Sizes** byelaw, which states that no person shall take from the fishery any fish of the following species (black seabream, brill, dab, conger eel, flounder, lemon sole, red mullet, shad, turbot, witch flounder) that measures less than the size listed when measured from the tip of the snout to the end of the tail. The minimum sizes contained within this byelaw differ from that in EU legislation.
- A separate Minimum Size Southern IFCA byelaw exists for Skates and Rays and this states that no person shall take any ray that measures less than 40 cm between the extreme tips of the wings or any wing which measures less than 20 cm in its maximum dimension and which is detached from the body of a skate or ray.
- A further Minimum size byelaw exists for **American hard-shelled clams** which states that no person shall remove from a fishery any clam of the species *Mercenaria mercenaria* which measures less than 63mm across the longest part of the shell.
- The **Fishing for Oysters, Mussles and Clams** byelaw states that the permitted methods of fishing for the aforementioned species are handpicking and dredging using a dredge with a ridged framed mouth.
- **Fishing for Cockles** must not take place in the Southern IFCA district between 1st February and 30th April. Cockle can only be fished for using handpicking, a rake or similar instrument, or with a dredge. Cockles which pass through a square gauge opening measuring 23.8mm along each side must not be removed from the fishery.
- Other regulations include minimum sizes, mesh sizes and catch composition as dictated by European legislation. European minimum sizes, listed under Council Regulation (EEC) 2019/1241 specify the minimum size for plaice is 27 cm and for bass is 42 cm. However, when certain gear types are used **The Landing Obligation** requires that specified bycatch species are retained at all sizes.

3.5 Table 7. Assessment of trawling and dredging on subtidal sand.

Feature	Attribute	Target	Potential pressure(s) and Associated Impacts	Likelihood of Impacts Occurring/Level of Exposure to Pressure	Current mitigation measures
Subtidal Sand	Distribution: presence and spatial distribution of biological communities; Structure: species composition of component communities; Structure and function: presence and abundance of key structural and influential species;	Maintain the presence and spatial distribution of subtidal sand communities [Maintain OR Recover OR Restore] the abundance of listed species*, to enable each of them to be a viable component of the habitat. Maintain the species composition of component communities.	Removal of non-target & target species, abrasion/ disturbance of the substrate on the surface of the seabed and penetration and/or disturbance of the substrate below the surface of the seabed, including abrasion were identified as potential pressures. Bottom towed gear can lead to the removal, damage or mortality of non-target & target species particularly epifaunal species, reduction in structural complexity and reduction in biodiversity and composition of benthic assemblages. Studies on the impacts of trawling and shellfish dredging in sandy habitats have reported reductions in abundance, biomass and species diversity, with undisturbed and lightly fished sites showing a greater abundance of epifauna. Other studies conducted in sandy habitats however have reported negligible impacts as a result of trawling disturbance. Benthic macrofauna in poorly sorted, gravelly or muddy sediments are reported to be more sensitive to trawling disturbance than well-sorted sandy sediments.	No trawling or dredging activity is known to occur within the Studland MCZ at present. There have been no sightings made within the site. However, it is known that in the past fishers who pump scoop in Poole Harbour trialled this equipment in the site. However, the trial was unsuccessful. Outside the site there are two trawl sightings in proximity to the site, seen between 2017 and 2019. These are located over 1km North East of Old Harry Rocks and another half a km off of sandbanks beach. Trawling is known to occur in Poole Bay, particularly along Bournemouth Beach. Only one vessel is however known to take part in the fishery. There is a lack of information surrounding the biotope and species present within the Studland MCZ. There is no post-survey site report, so information on the substrate type and associated communities is not available making it hard to ascertain site-specific impacts of bottom towed fishing gear on associated communities. The generic habitat description of subtidal sands indicates that they support communities including flat fish, sand eels, worms and bivalves, razor shells and sand mason worms. Scientific literature generally highlights that benthic communities associated with sand and muddy sand habitats can be vulnerable to the cumulative long-term bottom towed fishing gear disturbance	**Vessel Used in Fishing byelaw** prohibits commercial fishing vessels over 12 metres from the Southern IFCA district. The reduction in vessel size also restricts the type of gear that can be used, with vessels often using lighter towed gear. **Fishing for Cockles** must not take place in the Southern IFCA district between 1st February and 30th April. Cockle can only be fished for using handpicking, a rake or similar instrument, or with a dredge. Cockles which pass through a square gauge opening measuring 23.8mm along each side must not be removed from the fishery.

			The timescale for recovery after trawling and dredging disturbance largely depends on sediment type, associated fauna and rate of natural disturbance, and variation in recovery arises from characteristics specific to the site. Generally speaking, locations subject to high levels of natural disturbance, or habitats with more clean or coarse sediments, the associated fauna are likely to be adapted to withstand and recover from disturbance.	and subsequent negative changes can be observed across a number of community measures (abundance, biodiversity etc.). Hall *et al*. (2008) assessed the most relevant habitat type (subtidal stable muddy sands, sandy muds and muds and stable subtidal fine sands) to have low sensitivity to a single pass (per year) and light fishing intensity (1 to 2 times a month) with respect to all types of bottom towed gear. The lack of site-specific information on biotope and associated communities makes assessing the impacts of trawling & shellfish dredging disturbance difficult. Research into the effects of anchoring in Studland bays seagrass beds using sediment cores found that in fauna samples were dominated by polychaetes, oligochaetes, bivalves and amphipods. It showed that in areas of bare sand sediment (anchor or mooring chain scars) there were significantly less organisms (Collins et al. 2010). This study may indicate that the subtidal sands of the site contain fewer and less diverse infauna than the sediments found amongst the seagrass habitat. The presence of Zostera Marina combined with information from MARLIN (Marine Life Information Network) indicates that the sands in Studland bay may be clean, muddy fine sand, or sandy mud. A number of sub biotopes may be present within this biotope making it difficult to assess the impacts of BTFG specifically to the habitats in the site.	

				However, based on Hall et al. (2008) assessment that sand and muddy sand habitats have a low sensitivity to individual fishing events and low fishing activity, the knowledge that BTFG does not occur within the site itself and therefore there is none to very limited levels of disturbance, it is believed that BTFG will not pose a significant risk to the subtidal sand feature and therefore will not hinder the ability of the feature to maintain its 'maintain' general management approach (GMA). It is worth noting that in the absence of a condition assessment for the site, Natural England undertook a vulnerability assessment for each feature as a proxy for condition. This assessment considers the activities which take place in the site and determines the GMA for each feature. However, such an assessment is relatively generic and does not take into a number of site-specific factors.	
	Structure: sediment composition and distribution;	Maintain the distribution of sediment composition types across the feature	Abrasion/ disturbance of the substrate on the surface of the seabed and penetration and/or disturbance of the substrate below the surface of the seabed, including abrasion were identified as potential pressures. Physical impacts on the seabed from trawling and dredging include scraping and ploughing, creation of depressions, trenches, scouring and flattening of the seabed, sediment resuspension and changes in the	No trawling or dredging activity is known to occur within the Studland MCZ at present. There have been no sightings made within the site. However, it is known that in the past fishers who pump scoop in Poole Harbour trialled this equipment in the site. However, the trial was unsuccessful. Outside the site there are two trawl sightings in proximity to the site, seen between 2017 and 2019. These are located over 1km North East of Old Harry Rocks and another half a km off of sandbanks beach. Trawling is known to occur in Poole Bay, particularly along Bournemouth Beach. Only one vessel is however known to take part in the fishery.	Addressed above.

			vertical distribution of sediment layers. Studies on the effects of otter trawling and dredging in sand, gravel and variable habitats have revealed the activities can lead to the removal of fine sediments, increases in median grain size, removal of biogenic structures, moved or overturn stones and boulders, smooth the seafloor and exposed sediment/shell fragments. Otter boards can leave distinct grooves or furrows, up to 10 centimetres deep and 0.2 to 2 metres wide. The penetration depth of tickler chains on a beam trawl can be up to 6 cm. The depth of such marks on the seafloor depend on the nature of the substrate, and are less in areas of coarser sediments. Intertidal shellfish dredging can leave furrows tens of centimetres deep, and can remain visible for day to weeks. Physical recovery of sediments largely depends on sediment type and energy regime. In high energy environments physical recovery can take days, whereas recovery in low energy environments can take months. Recovery periods for sandy	The lack of site-specific information on sediment composition makes assessing the impacts of BTFG disturbance difficult. The habitat feature map for the site shows the entire site is dominated by subtidal sands, however this data is based on UK Sea Map 2018, which has a low resolution of only 100m. Research into the effects of anchoring in Studland bays seagrass beds using sediment cores found that in most samples mean grain size indicated the sediment was medium sand, with low (<7%) silt fraction (Collins et al. 2010). However, samples were taken from within a mixture of seagrass habitats and anchor or mooring disturbed areas which could have affected the sediment composition of the samples. The presence of Zostera Marina combined with information from MARLIN (Marine Life Information Network) indicates that the sands in Studland bay may be clean or muddy fine sand, or sandy mud. A number of sub biotopes may be present within his biotope making it difficult to assess the impacts of BTFG specifically to the habitats in the site. However, based on Hall et al. (2008) assessment that sand and muddy sand habitats have a low sensitivity to individual fishing events and low fishing activity, the knowledge that BTFG does not occur within the site itself and therefore there is no to very limited levels of disturbance, and the suggested low silt content within the sediments (Collins et al., 2010) it is believed that BTFG will not pose a significant risk to the subtidal sand feature and therefore will not hinder the ability of	

			sediments is likely to be days to months.	the feature to maintain its 'maintain' general management approach (GMA). It is worth noting that in the absence of a condition assessment for the site, Natural England undertook a vulnerability assessment for each feature as a proxy for condition. This assessment considers the activities which take place in the site and determines the GMA for each feature. However, such an assessment is relatively generic and does not take into a number of site-specific factors.	
Subtidal sand	Supporting processes: water quality - turbidity	Maintain natural levels of turbidity (eg concentrations of suspended sediment, plankton and other material) across the habitat.	Smothering and siltation rate changes (Light) and Changes in suspended solids (water clarity) were identified as potential pressures. The resuspension of sediment can impact upon benthic communities through smothering, burial and increased turbidity. These effects may extend to organisms living a distance away from the fished area. The timescale for recovery after trawling disturbance largely depends on sediment type, associated fauna and rate of natural disturbance, and variation in recovery arises from characteristics specific to the site. Generally speaking, locations subject to high levels of natural disturbance, the associated fauna are likely to be adapted to withstand and recover from disturbance.	No trawling or dredging activity is known to occur within the Studland MCZ at present. There have been no sightings made within the site. However, it is known that in the past fishers who pump scoop in Poole Harbour trailed this equipment in the site. However, the trial was unsuccessful. Outside the site there are two trawl sightings in proximity to the site, seen between 2017 and 2019. These are located over 1km North East of Old Harry Rocks and another half a km off of sandbanks beach. Trawling is known to occur in Poole Bay, particularly along Bournemouth Beach. Only one vessel is however known to take part in the fishery. There is a lack of information surrounding the biotope and species present within the Studland MCZ. There is no post-survey site report, so information on the substrate type and associated communities is not available making it hard to ascertain site-specific impacts of bottom towed fishing gear on associated communities.	

				The generic habitat description of subtidal sands indicates that they support communities including flat fish, sand eels, worms and bivalves, razor shells and sand mason worms. Research has found that high levels of sediment and regular exposure can cause sever impacts. Increased turbidity can inhibit respiratory and feeding functions of benthic organisms, and cause hypoxia or anoxia. Small organisms and immobile species are particularly vulnerable to smothering. The severity of the impact is determined by sediment type, the level of sediment burden and the sensitivity of organisms which is largely related to their biology (i.e. size, relationship to substrate, life history, mobility). Hall *et al*. (2008) assessed the most relevant habitat type (subtidal stable muddy sands, sandy muds and muds and stable subtidal fine sands) to have low sensitivity to a single pass (per year) and light fishing intensity (1 to 2 times a month) with respect to all types of bottom towed gear. Tillin et al. (2010) assessed the sensitivity of these habitats to changes in siltation and found subtidal sand to have a medium sensitivity. Research into the effects of anchoring in Studland bays seagrass beds using sediment cores found that in most samples mean grain size indicated the sediment was medium sand, with low (<7%) silt fraction (Collins et al. 2010). However, samples were taken from within a mixture of seagrass habitats and anchor or mooring disturbed areas which could have affected the sediment composition of the samples.	

				However, based on Hall et al. (2008) assessment that sand and muddy sand habitats have a low sensitivity to individual fishing events and low fishing activity, the knowledge that BTFG does not occur within the site itself and therefore there is no to very limited levels of disturbance, and the suggested low silt content within the sediments (Collins et al., 2010) it is believed that the pressure siltation rate changes will not pose a significant risk to the subtidal sand feature and therefore will not hinder the ability of the feature to maintain its 'maintain' general management approach (GMA). It is worth noting that in the absence of a condition assessment for the site, Natural England undertook a vulnerability assessment for each feature as a proxy for condition. This assessment considers the activities which take place in the site and determines the GMA for each feature. However, such an assessment is relatively generic and does not take into a number of site-specific factors.	
Seagrass Beds	Distribution: presence and spatial distribution of biological communities; Structure and function: presence and abundance of key structural and influential species;	Maintain the presence and spatial distribution of subtidal seagrass bed communities [Maintain OR Recover OR	Removal of non-target species, abrasion/ disturbance of the substrate on the surface of the seabed and penetration and/or disturbance of the substrate below the surface of the seabed, including abrasion were identified as potential pressures. Bottom towed gear can lead to the removal, damage or mortality of non-target & target species particularly epifaunal species, reduction in	No trawling or dredging activity is known to occur within the Studland MCZ at present. There have been no sightings made within the site. However, it is known that in the past fishers who pump scoop in Poole Harbour trialled this equipment in the site. However, the trial was unsuccessful. Outside the site there are two trawl sightings in proximity to the site, seen between 2017 and 2019. These are located over 1km North East of Old Harry Rocks and another half a km off of sandbanks beach. Trawling is known to occur in Poole Bay, particularly along Bournemouth	

	Structure: species composition of component communities; Structure: biomass; Structure: rhizome structure and reproduction	Restore] the abundance of listed species*, to enable each of them to be a viable component of the habitat. Recover the species composition of component communities.	structural complexity and reduction in biodiversity and composition of benthic assemblages. Studies on the impacts of trawling and shellfish dredging in sandy habitats have reported reductions in abundance, biomass and species diversity, with undisturbed and lightly fished sites showing a greater abundance of epifauna. Other studies conducted in sandy habitats however have reported negligible impacts as a result of trawling disturbance. The timescale for recovery after trawling and dredging disturbance largely depends on sediment type, associated fauna and rate of natural disturbance, and variation in recovery arises from characteristics specific to the site. Generally speaking, locations subject to high levels of natural disturbance, or habitats with more clean or coarse sediments, the associated fauna are likely to be adapted to withstand and recover from disturbance.	Beach. Only one vessel is however known to take part in the fishery. There is a lack of information surrounding the biotope and species present within the Studland MCZ. There is no post-survey site report, so information on the substrate type and associated communities is not available making it hard to ascertain site-specific impacts of bottom towed fishing gear on associated communities. The generic description of seagrass beds identifies that the habitat provides food for waterfowl, and nursery habitats for juvenile fish. Other animals known to be associated with seagrass beds include seahorses, anemones, crabs, and other shellfish. Research into the effects of anchoring in Studland bays seagrass beds using sediment cores found that in fauna samples were dominated by polychaetes, oligochaetes, bivalves and amphipods. Studies indicate that trawling and dredging in seagrass habitats leads to significant reduction in seagrass biomass, shoot density and canopy cover. Due to this the biological assemblages of the beds are altered, as are sediment compositions and structure. Literature indicates that recovery of seagrass to these impacts can take in excess of 10 years. Hall et al., 2008 assessed the sensitivity of seagrass bed to all bottom towed fishing gear types at all fishing intensity levels to be high.	

				At the present time trawling and dredging activities are not known to occur within or in close proximity to the seagrass habitats of Studland MCZ. However, the habitat is highly sensitive to these fishing gear types and has a long recovery period. If fishing were to occur over the habitat it would lead to the instant removal of the feature. Therefore, it is believed that trawling & dredging will pose a significant risk to the seagrass beds in the MCZ, and could therefore hinder the ability of the feature to achieve its 'recover' general management approach (GMA). It is worth noting that in the absence of a condition assessment for the site, Natural England undertook a vulnerability assessment for each feature as a proxy for condition. This assessment considers the activities which take place in the site and determines the GMA for each feature. However, such an assessment is relatively generic and does not take into a number of site-specific factors.	
	Extent and distribution; Structure: sediment composition and distribution;	Recover the total extent and spatial distribution of seagrass beds. Maintain the distribution of sediment composition types across the	Addressed above	Addressed above	Addressed above

		feature/subfeature.			
	Supporting processes: light levels; Supporting processes: water quality - turbidity	Maintain the natural light availability to the seagrass bed. Maintain natural levels of turbidity (e.g. concentrations of suspended sediment, plankton and other material) across the habitat.	Smothering and siltation rate changes (Light) and Changes in suspended solids (water clarity) were identified as potential pressures. The resuspension of sediment can impact upon benthic communities through smothering, burial and increased turbidity. These effects may extend to organisms living a distance away from the fished area. The timescale for recovery after trawling disturbance largely depends on sediment type, associated fauna and rate of natural disturbance, and variation in recovery arises from characteristics specific to the site. Generally speaking, locations subject to high levels of natural disturbance, the associated fauna are likely to be adapted to withstand and recover from disturbance.	No trawling or dredging activity is known to occur within the Studland MCZ at present. There have been no sightings made within the site. However, it is known that in the past fishers who pump scoop in Poole Harbour trialled this equipment in the site. However, the trial was unsuccessful. Outside the site there are two trawl sightings in proximity to the site, seen between 2017 and 2019. These are located over 1km North East of Old Harry Rocks and another half a km off of sandbanks beach. Trawling is known to occur in Poole Bay, particularly along Bournemouth Beach. Only one vessel is however known to take part in the fishery. There is a lack of information surrounding the biotope and species present within the Studland MCZ. There is no post-survey site report, so information on the substrate type and associated communities is not available making it hard to ascertain site-specific impacts of bottom towed fishing gear on associated communities. The generic description of seagrass beds identifies that the habitat provides food for waterfowl, and nursery habitats for juvenile fish. Other animals known to be associated with seagrass beds include seahorses, anemones, crabs, and other shellfish. Research has found that high levels of sediment and regular exposure can cause sever impacts. Increased turbidity can inhibit respiratory and	

				feeding functions of benthic organisms, and cause hypoxia or anoxia. Small organisms and immobile species are particularly vulnerable to smothering. The severity of the impact is determined by sediment type, the level of sediment burden and the sensitivity of organisms which is largely related to their biology (i.e. size, relationship to substrate, life history, mobility). Hall et al., 2008 assessed the sensitivity of seagrass bed to all bottom towed fishing gear types at all fishing intensity levels to be high. Tillin et al. (2010) assessed the sensitivity of these habitats to changes in siltation and found seagrass beds to have a high sensitivity. At the present time trawling and dredging activities are not known to occur within or in close proximity to the seagrass habitats of Studland MCZ. Therefore, it is believed that trawling & dredging will not pose a significant risk of increasing siltation rate and therefore this pressure will not hinder the ability of the feature to achieve its ‘recover’ general management approach (GMA). It is worth noting that in the absence of a condition assessment for the site, Natural England undertook a vulnerability assessment for each feature as a proxy for condition. This assessment considers the activities which take place in the site and determines the GMA for each feature. However, such an assessment is relatively generic and does not take into a number of site-specific factors.	

4 Proposed management measures

In recognition of the potential pressures of bottom towed fishing gear (particularly trawling and shellfish dredging) upon designated features and their supporting habitats, Southern IFCA will follow the process of introducing permanent bottom towed fishing gear (BTFG) closure areas in order to protect seagrass beds in the Studland MCZ. It was found that trawling and dredging are likely to pose a significant risk to the achievement of general management approach of the feature and conservation objectives of the site.

The bottom towed fishing gear closed areas have been chosen based on seagrass presence data provided by Natural England. The bottom towed gear fishing closure areas are designed to fully protect seagrass beds against BTFG, by completely prohibiting all types of bottom towed fishing, including trawling and dredging, over the seagrass within the site. Each area has been designed to incorporate a buffer around the seagrass feature data. The buffer distance is determined by the following formula: Deepest feature depth * 4 +10m. The buffer ensures that if fishing was to occur along the line of the closed area, the actual trawl/dredge location would not occur over the feature itself.

The measures presented are draft and used to illustrate protection based purely on location. When developing management other evidence such as fishing activity and consultation with the local community may feed into the development of spatial closed areas.

Management will be introduced in the upcoming update to the Southern IFCA Bottom towed Fishing Gear Byelaw 2016. The primary reason for management options is to protect seagrass beds, which are known to be highly sensitive to BTFG against the impacts caused by bottom towed fishing gear.

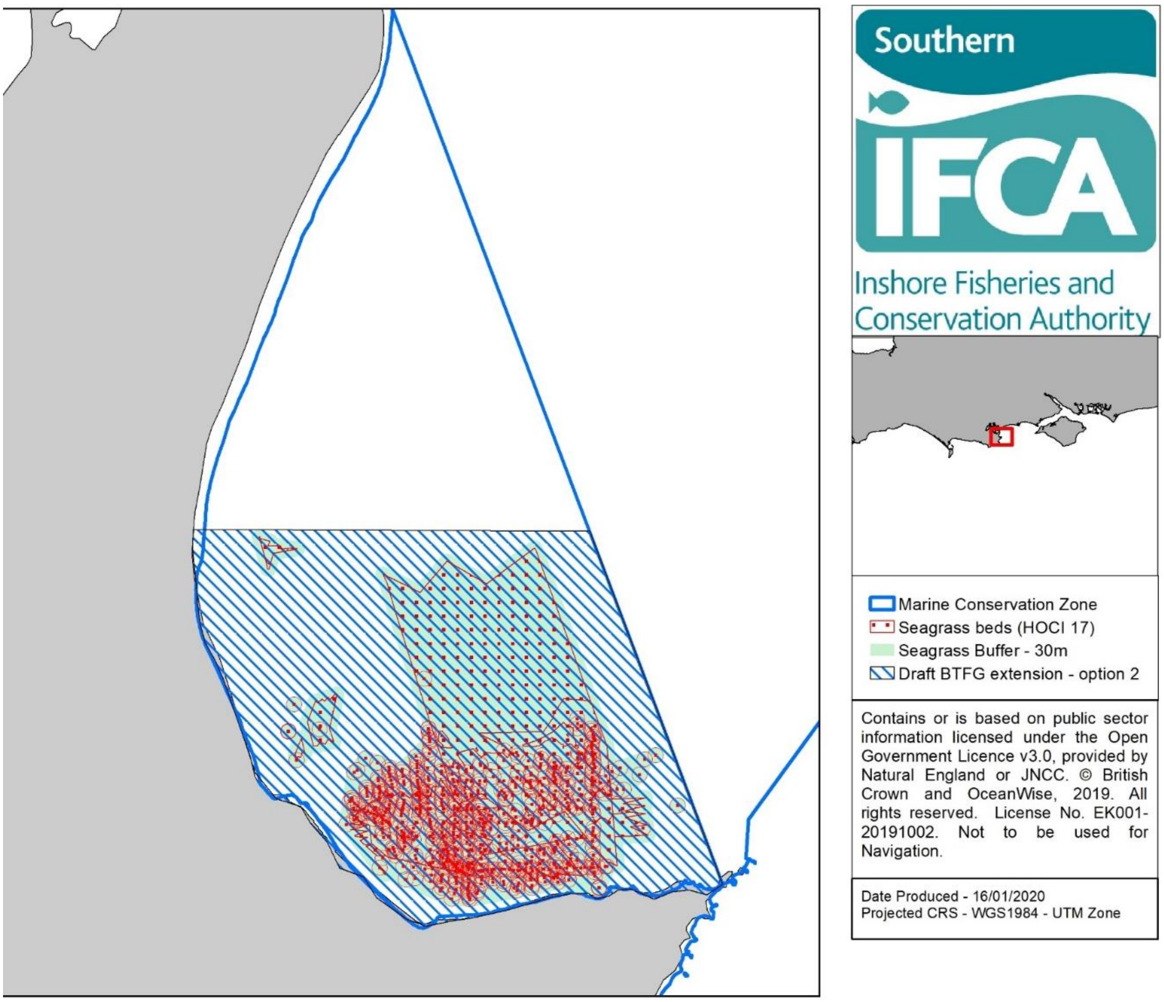


Figure 8 Draft Bottom Towed Fishing Gear closed area in the Studland MCZ to protect seagrass beds

5 Conclusion

In order to conclude whether types of bottom towed fishing gear (trawls and shellfish dredges) pose a significant risk, it is necessary to assess whether the impacts of the activities will hinder the achievement of the general management approach of the designated feature (seagrass beds) of 'recover to favourable condition' and (subtidal sands) to achieve its 'maintain at favourable condition' and the sites conservation objectives, namely:

"The conservation objective of each of the zones is that the protected habitats:

1. are maintained in favourable condition if they are already in favourable condition

2. be brought into favourable condition if they are not already in favourable condition

For each protected feature, favourable condition means that, within a zone:

1. its extent is stable or increasing

2. its structure and functions, its quality, and the composition of its characteristic biological communities (including diversity and abundance of species forming part or inhabiting the habitat) are sufficient to ensure that its condition remains healthy and does not deteriorate

Any temporary deterioration in condition is to be disregarded if the habitat is sufficiently healthy and resilient to enable its recovery.

The review of the research into the impacts of bottom towed fishing gear on subtidal sands (subtidal stable muddy sands, sandy muds and muds, Stable subtidal fine sands) reported the habitat to have low sensitivity to a single pass and light fishing activity (1-2 times per month). Natural disturbance within the site was found to be low. Therefore, it is concluded that the level of fishing activity in and outside of the site will not prevent the ability of subtidal sand to attain their 'maintain' general management approach.

The review of the research into the impacts of bottom towed fishing gear on seagrass beds reported the habitat to have a high sensitivity to all levels of fishing activity. One single pass of dredge fishing gear could entirely remove the feature. Therefore, it is concluded that the fishing activity will prevent the ability of seagrass beds to attain their 'recover' general management approach.

Having reviewed a wide range of evidence, including scientific literature, IFCO knowledge, habitat feature mapping (including bathymetric data), it was concluded that bottom towed fishing gear is likely to pose a significant risk to seagrass beds within the Studland Bay MCZ. The rationale for this conclusion is summarised below:

- IFCO knowledge indicates that trawling and shellfish dredging do not occur within the site, but trawling occurs outside of the site. Shellfish dredging was trailed in the site a number of years ago but was unsuccessful.
- No sightings of the activities have been made in the site.
- A review of scientific literature demonstrated that bottom towed fishing gear at any intensity can lead to the direct removal mortality of non-target & target species particularly seagrass itself. Additionally, bottom towed fishing gear can lead to physical disturbance of the seabed including creation of furrows and mixing of sediment layers. However, it is important to note that few studies have been completed.
- Sensitivity of seagrass habitats to pressures associated with trawls and dredges at all intensities is high.
- Recovery of seagrass habitats are predicted to be 10+ years.

It is therefore recognised that the activities have the potential to pose a significant risk upon the seagrass beds attributes:

- Extent and distribution
- Structure: sediment composition and distribution
- Distribution: presence and spatial distribution of biological communities

- Structure and function: presence and abundance of key structural and influential species
- Structure: species composition of component communities
- Structure: rhizome structure and reproduction
- Structure: biomass

Upon the provision of additional evidence, including conservation advice for the site, and up to date habitat maps, Southern IFCA feel it is now appropriate for refinement to the spatial extent of the current closures and inclusion of additional bottom towed fishing gear closed areas. This is to support the general management approach to ‘recover’ the seagrass beds to a favourable condition. The primary reason for management is to protect the seagrass beds habitat feature.

In summary, when the above evidence, fishing activity levels, current and proposed management measures are considered it has been concluded that bottom towed fishing gear will not pose a significant risk to the achievement of sites conservation objectives to ‘recover’ seagrass beds and to ‘maintain’ subtidal sand to favourable condition. Southern IFCA must seek to ensure that the conservation objectives of any MCZ in the district are furthered.

6 In-combination assessment

6.1 Other fishing activities

Fishing activity	Potential for in-combination effect
Static – pots/traps (Pots/creels – crustacean & cuttle pots)	Potting for crab and lobster takes place over rocky substrate and will therefore not overlap with trawling activity which takes place over subtidal sediments. Potting in general is also considered to be low impact (Grieve *et al.,* 2014) and not likely to lead to any in-combination effects. In addition, static gear types such as potting and mobile gear types such as trawling are not compatible and so often occur in different areas, thus largely eliminating any spatial overlap between the two.
Static – fixed & passive nets (Gill nets, trammels, entangling, drift nets)	It is anticipated that static fixed nets are used within the site in areas of shallow water and will therefore not overlap with trawling activity. Netting is also a low impact activity and not likely to lead to any in-combination effects. In addition, static gear types such as netting and mobile gear types such as trawling are not compatible and so often occur in different areas, thus largely eliminating any spatial overlap between the two.
Lines (Longlines – demersal, Handlines)	It is anticipated that demersal longlines and handlines are used within the site. The area where the activity may take place however is unknown. Demersal longlining and handline are low impact activities and not likely to lead to any in-combination effects. In addition, static gear types such as longlining and mobile gear types such as trawling are not compatible and so often occur in different areas, thus largely eliminating any spatial overlap between the two.
Intertidal work (Hand work and digging with forks)	Intertidal hand work leads to the same pressures as trawling and shellfish dredging. Trawling does not occur in the intertidal area so there will be no in-combination effect from this activity. Shellfish dredging can occur in intertidal areas therefore the activities could overlap. However, shellfish dredging does not occur within the site, and will be permanently prohibited over the entirety of the seagrass beds. Therefore, there will be no in-combination effect. The effects of intertidal work on seagrass beds has been assessed in a separate MCZ assessment.

6.2 Plans/projects

Consultation with Natural England did not recognise any plans or projects which had the possibility to lead to in-combination effect with fishing activities.

7 References

Ardizonne, G.D., Tucci, P., Somaschini, A. & Belluscio, A., 2000. Is bottom trawling partly responsible for the regression of *Posidonia oceania* meadows in the Mediterranean Sea? In: Kaiser, M.J. & de Groot, S. J. (eds) *Effects of fishing on on-target species and habitats*. Blackwell Science, London. Pp 37 - 46

Auster, P.J., R.J. Malatesta, R.W. Langton, L. Watling, P.C. Valentine, C.L.S. Donaldson, E.W. Langton, A.N. Shepard, & I.G. Babb. 1996. The impacts of mobile fishing gear on seafloor habitats in the Gulf of Maine (northwest Atlantic): implications for conservation of fish populations. *Rev. Fish. Sci*., **4 (2):** 185-202.

Ball, B., Munday, B., & Tuck, I.D. 2000. Effects of otter trawling on the benthos and environment in muddy sediments. In Kaiser, M.J. & de Groot, S.J. (Eds). *The Effects of Fishing on Non-target Species and Habitats*. Blackwell Science. pp. 69-82

Bergman, M.J.N, Fonds, M., Hup, M. & Stam, A. 1990. Direct effects of beam trawl fishing on benthic fauna in the North Sea. ICES C.M. MINI:11.

Bergman, M.J.N. & van Santbrink, J.W. 2000. Mortality in megafaunal benthic populations caused by trawl fisheries on the Dutch continental shelf in the North Sea in 1994. *ICES J. Mar. Sci*., **57**: 1321-1331.

Bolam, S.G., Coggan, R.C., Eggleton, J., Diesing, M. & Stephens, D. 2014. Sensitivity of microbenthic secondary production to trawling in the English sector of the Greater North Sea: A biological trait approach. *J. Sea Res*., **85**: 162-177.

Boulcott, P., & Howell, T.R.W., 2011. The impact of scallop dredging on rocky-reef substrata. *Fisheries Research.* **110:**415-420.

Boulcott, P., Millar, C.P. and Fryer, R.J. 2014. Impact of scallop dredging on benthic epifauna in a mixed-substrate habitat. *ICES Journal of Marine Science*. **71(4):**834-844. doi.10.1093/icesjms/fst197

Bridger, J. P. 1972. Some observations on the penetration into the sea bed of tickler chains on a beam trawl. ICES CM B:7, 9 pp.

Brylinsky, M., Gibson, J. & Gordon, D.C. 1994. Impacts of flounder trawls on the intertidal habitat and community of the Minas Basin, Bay of Fundy. *Can. J. Fish Aquat. Sci*., **51:** 650-61.

Caddy, J.F., 1973. Underwater observations on tracks of dredges and trawls and some effects of dredging on scallop ground. *Journal of the Fisheries Research Board of Canada.* **30 (2):** 173-180

Callaway, R., Engelhard, G.H., Dann, J., Cotter, J. & Rumohr, H. 2007. A century of North Sea epibenthos and trawling: comparison between 1902–1912, 1982–1985 and 2000. *Mar. Ecol. Prog. Ser*., **346**, 27-43.

Coen, L.D. 1995. A review of the potential impacts of mechanical harvesting on subtidal and intertidal shellfish resources. SCDNR-MRRI, 46 pp.

Collie, J.S., G.A. Escanero, and P.C. Valentine. 1997. Effects of bottom fishing on the benthic megafauna of Georges Bank. *Mar. Ecol. Prog. Ser*., **155:**159-172.

Collie, J.S., Hall, S.J., Kaiser, M.J. & Poiner, I.R. 2000. A quantitative analysis of fishing impacts on shelf-sea benthos. *J. Anim. Ecol*., **69:** 785-798.

Currie, D. R., and Parry, G. D. 1996. Effects of scallop dredging on a soft sediment community: a large-scale experimental study. Marine Ecology Progress Series, **134:** 131–150.

Curtis J.M.R. and Vincent A.C.J., 2006. Life history of an unusual marine fish: survival, growth and movement patterns of Hippocampus guttulatus Cuvier 1829, 2006. *Journal of Fish Biology.* **68:** 707-733.

Davison, D.M. and Hughes, D.J. 1998. Zostera biotopes: An overview of dynamics and sensitivity characteristics for conservation management of marine SACs, Scottish Association for Marine Science, (UK Marine SACs Project). **1:** 40-41.

De Groot, S.J. & Lindeboom, H.J. 1994. Environmental impact of bottom gears on benthic fauna in relation to natural resources management and protection of the North Sea. Texel, Netherlands. Netherlands Institute for Sea Research.

De Jonge, V.N. & De Jonge, D.J. 1992. Role of tide, light and fisheries in the decline of Zostera marina L. in the Dutch Wadden Sea. Netherlands Institute for Sea Research Publication. **20:** 161-76.

DeAlteris, J., Skrobe, L. & Lipsky, C. 1999. The significance of seabed disturbance by mobile fishing gear relative to natural processes: a case study in Narragansett Bay, Rhode Island. In Benaka, L (Ed). *Fish habitat: essential fish habitat and rehabilitation*. American Fisheries Society, Symposium 22, Bethesda, Maryland, pp. 224-237

Depestele, J., Courtens, W., Degraer, S., Haelters, J., Hostens, K., Houziaux, J.S., Merckz, B., Polet, H., Rabaut, M., Stienen, E.W.M., Vandendriessche, S., Verfaillie, E. & Vincx, M. 2012. An integrated impact assessment of trammel net and beam trawl fisheries "WAKO II" - Final Report. Project SD/NS/O8A. Brussels: Belgian Science Policy Office. 234 pp.

Dernie, K.M., Kaiser, M.J. & Warwick, R.M. 2003. Recovery rates of benthic communities following physical disturbance. *J. Anim. Ecol*., **72**, 1043-1056.

Dorsey, E.M., ad Pederson, J. 1998. Effects of Fishing Gear on the Sea Floor of New England. Conservation Law Foundation. Available at: http://nsgl.gso.uri.edu/mit/mitw97003/effects_of_fishing_gear.htm

Dorsey, E.M., ad Pederson, J. 1998. Effects of Fishing Gear effects of dredging on a scallop ground. J. Fish. Res. Board Can. **30:** 173–180.

Engel, J. & Kvitek, R. 1998. Effects of otter trawling on benthic community in Monterey Bay National Marine Sanctuary. *Cons. Biol*., **12(6):** 1204-214.

Foden, J., Rogers, S.I. & Jones, A.P. 2010. Recovery of UK seabed habitats from benthic fishing and aggregate extraction—towards a cumulative impact assessment. *Marine ecological progress series*. **411**:259-270.

Fonseca, M.S., Thayer, G.W., Chester, A.J. & Foltz, C. 1984. Impact of Scallop Harvesting on Eelgrass (Zostera marina) Meadows: Implications for Management. *North American Journal of Fisheries management.* **4 (3):** 286-293

Fonteyne, R. 2000. Physical impact of beam trawls on seabed sediments. In Kaiser, M.J. & de Groot, S.J. (Eds). *The Effects of Fishing on Non-target Species and Habitat*. Blackwell Science. pp. 15-36

Freeman, S. M., Richardson, C.A. & Seed, R. 2001. Seasonal abundance, spatial distribution, spawning and growth of *Astropecten irregularis* (Echinodermata: Asteroidea). *Estuar. Coast. Shelf. Sci*., **53:** 39–49.

Freese, L., Auster, P. J., Heifetz, J. & Wing, B. L. 1999. Effects of trawling on seafloor habitat and associated invertebrate taxa in the Gulf of *Alaska. Mar. Ecol. Prog. Ser*., **182:** 119-126.

Gilkinson, K., Paulin, M., Hurley, S. & Schwinghamer, P. 1998. Impacts of trawl door scouring on infaunal bivalves: results of a physical trawl door model/dense sand interaction. *J. Exp. Mar. Biol. & Ecol*., **224:** 291-312.

Gonzalez-Correa, J.M., Bayle, J.T., Sanchez-Lizaso, J.L. & Valle, C., Sanchez-Jerez, P. Ruiz, J.M. 2005. Recovery of deep *Posidonia oceanica* meadows degraded by trawling. *Journal of Experimental Marine Biology and Ecology*. **320:** 65–76

Goodchild, R., Brutto, D., Snaith, E., Frost, N., Kaiser, M. & Salmon, P. 2015. Analysis of existing data to study effects of towed fishing gears on mobile sediments against a background of natural variability. Funded by Department for Environment, Food and Rural Affairs (Defra). 65 pp.

Goss-Custard, J.D. 1977. The ecology of the Wash. III. Density-related behaviour and the possible effects of a loss of feeding grounds on wading birds (Charadrii). *J. Anim. Ecol*., **14:** 721-739.

Grieve, C., Brady, D.C. & Polet, H. 2014. Best practices for managing, measuring and mitigating the benthic impacts of fishing – Part 1. *Marine Stewardship Council Science Series*. **2:** 18 – 88.

Gubbay, S. & Knapman, P.A. 1999. A review of the effects of fishing within UK European marine sites. UK Marine SACs Project. 134 pp.

Hall, K., Paramor, O.A.L., Robinson, L.A., Winrow-Giffin, A., Frid, C.L.J., Eno, N.C., Dernie, K.M., Sharp, R.A.M., Wyn, G.C. & Ramsay, K. 2008. Mapping the sensitivity of benthic habitats to fishing in Welsh Waters: development of a protocol. CCW (Policy Research) Report No: 8/12. 85 pp.

Hiddink, J.G., Jennings, S., Kaiser, M.J., Queirós, A.M., Duplisea, D.E. & Piet, G.J. 2006a. Cumulative impacts of seabed trawl disturbance on benthic biomass, production and species richness in different habitats. *Can. J. Fish. Aquat. Sci*., **63**: 721-736.

Hinz, H., Murray, L.G., Malcolm, F.R. & Kaiser, M.J. 2012. The environmental impacts of three different queen scallop (Aequipecten opercularis) fishing gears. *Marine Environmental Research*. **73**: 85-95.

Hinz, H., Prieto, V. & Kaiser, M.J. 2009. Trawl disturbance on benthic communities: chronic effects and experimental predictions. *Ecol. Appl*., **19 (3):** 761-773.

Howell, B.R. & Shelton, R.G.J. 1970. The effect of china clay on the bottom fauna of St Austell and Mevagissey Bays. *J. Mar. Biol. Assoc*. U. K., **50 (3**): 593-607.

Humborstad, O.-B., Nøttestad, L., Løkkeborg, S. & Rapp, H.T. 2004. RoxAnn bottom classification system, side scan sonar and video-sledge: spatial resolution and their use in assessing trawling impacts. *ICES J. Mar. Sci*., **61**: 53-63.

ICES. 1992. Report of the study group on ecosystem effects of fishing activities. ICES C.M.1992/G:11.

Jennings, S. & Kaiser, M.J., 1998. The effects of fishing on marine ecosystems. *Adv. Mar. Biol*., **34**, 201–352.

Jennings, S., Dinmore, T.A., Duplesea, D.E., Warr, K.J. & Lancaster, J.E. 2001. Trawling disturbance can modify benthic production processes. *J. Anim. Ecol*., **70:** 459–475.

Jennings, S., M.D. Nicholson, T.A. Dinmore & J. Lancaster, 2002. Effects of chronic trawling disturbance on the production of infaunal communities. *Mar. Ecol. Prog. Ser*., **243**: 251–260.

Jones, J.B. 1992. Environmental impact of trawling on the seabed: a review. *New Zeal. J. Mar. Freshwat. Res*., **26**: 59-67.

Kaiser, M.J. & Spencer, B.E. 1996. The effects of beam-trawl disturbance on infaunal communities in different habitats. *J. Anim. Ecol*., **65:** 348-58.

Kaiser, M.J. Unpublished. A summary of the impacts of scallop dredging on seabed biota and habitats. *DRAFT NOT TO BE DISTRIBUTED BEYOND DSFC, SSFC, WTs and EN*

Kaiser, M.J., Cheney, K., Spence, F.E., Edwards, D.B. & Radford, K. 1999. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure. VII. The effects of trawling disturbance on the fauna associated with the tubeheads of serpulid worms. *Fish. Res*., **40**: 195-205.

Kaiser, M.J., Clarke, K.R., Hinz, H., Austen, M.C.V., Somerfield, P.J. & Karakassis, I. 2006. Global analysis of response and recovery of benthic biota to fishing. *Mar. Ecol. Prog. Ser*., **311:** 1-14.

Kaiser, M.J., Collie, J.S., Hall, S.J., Jennings, S. & Poiner, I.R. 2002. Modification of marine habitats by trawling activities: prognosis and solutions. *Fish and Fisheries*, **3:** 1-24.

Kaiser, M.J., D.B. Edwards & Spencer, B.E. 1996. Infaunal community changes as a result of commercial clam cultivation and harvesting. *Aquat. Living Resour*., **9:** 57-63.

Kaiser, M.J., D.B. Edwards & Spencer, B.E. 1996B. Infaunal community changes as a result of commercial clam cultivation and harvesting. *Aquat. Living Resour*., **9:** 57-63.

Kaiser, M.J., Edwards, D.B., Armstrong, P.J., Radford, K., Lough, N.E.L., Flatt, R.P. & Jones, H.D. 1998. Changes in megafaunal benthic communities in different habitats after trawling disturbance. *ICES J. Mar. Sci*., **55**: 353-361.

Kaiser, M.J., Ramsay, K., Richardson, C.A., Spence, F.E., Brand, A.R. 2000. Chronic fishing disturbance has changed shelf sea benthic community structure. *J. Anim. Ecol*., **69:** 494–503.

Kenchington, E. L. R., Prena, J., Gilkinson, K. D., Gordon Jr, D. C., Macisaac, K., Bourbonnais, C., Schwinghamer, P. J., Rowell, T. W., McKeown, D. L. & Vass, W. P., 2001. Effects of experimental otter trawling on the macrofauna of a sandy bottom ecosystem on the Grand Banks of Newfoundland. *Can. J. Fish. Aquat. Sci*., **58(6):** 1043-1057.

Kyte, M.A. & Chew, K.K. 1975. A review of the hydraulic escalator shellfish harvester and its known effects in relation to the soft-shell clam, *Mya arenaria*. Seattle (WA) Washington Sea Grant Program, University of Washington. 32 pp.

Lawson, J.M., Foster, S.J. & Vincent, A.C.J. 2017. Low bycatch rates add up to big numbers for a genus of small fishes. *Fisheries*. **42(1):** 19-33

Leth J.O. & Kuijpers A. 1996. Effects on the seabed sediment from beam trawling in the North Sea. ICES 1996. Annual Science Conference. Mini-symposium: “Ecosystem Effects of Fisheries”. ICES C.M. 1996/Mini 3.

Lindeboom, H.J. & S.J. de Groot, 1998. Impact II. The effects of different types of fisheries on the North Sea and Irish Sea bent hic ecosystems. NIOZ Rapport 1998-1. 404 pp.

Lindholm, J., Gleason, M., Kline D., Clary, L., Rienecke, S., Bell, M. & Kitaguchi, B. 2013. Central Coast Trawl Impact and Recovery Study: 2009-2012 Final Report. Report to the California Ocean Protection Council. 49 pp.

Løkkeborg, S. 2005. Impacts of trawling and scallop dredging on benthic habitats and communities. FAO Fisheries Technical Paper 472. Food and Agriculture Organisation of the United Nations. 69 pp.

Malecha, P.W., & Stone, R.P. 2009. Response of the sea whip *Halipteris willemoesi* to simulated trawl disturbance and its vulnerability to subsequent predation. *Marine ecological progress series*. **388:**197-206

Manning, J.H. 1957. The Maryland softshell clam industry and its effects on tidewater resources. Md. Dep. Res. Educ. Resour. Study Rep. **11:** 25 pp.

McConnaughey, R.A., Mier, K.L. & Dew, C.B. 2000. An examination of chronic trawling on soft bottom benthos of the eastern Bering Sea. *ICES J. Mar. Sci*., **57:** 1388-1400.

Mercaldo-Allen, R. & Goldberg, R. 2011. Review of the Ecological Effects of Dredging in the Cultivation and Harvest of Molluscan Shellfish. NOAA Technical Memorandum NMFS-NE-220. 84 pp.

Moore and Jennings, 2000 (ed). Commercial fishing: The wider ecological impacts.

Moran, M.J. & Stephenson, P.C. 2000. Effects of otter trawling on macrobenthos and management of demersal scale fish fisheries on the continental shelf of north-western Australia. *ICES J. Mar. Sci*., **57:** 510-516.

Morgan, L.E. & Chuenpagdee, R. 2003. *Shifting gears: Addressing the collateral impacts of fishing methods in US waters*. PEW Science Series. Washing D.C., Island Press.

Neckles, H.A., Short, F.T., Barker, S. and Kopp. B.S. 2005. Disturbance of eelgrass Zostera marine by commercial mussel Mytilus edulis harvesting in Maine: dragging impacts and habitat recovery. *Marine Ecology Progress Series***.** **285:** 57-73.

Nejrup, L.B. and Pedersen, M.F. 2008. Effects of salinity and water temperature on the ecological performance of Zostera marina. *Aquatic Botany*. **88:** 239-246.

Nilsson, H.C. & Rosenberg, R. 2003. Effects on marine sedimentary habitats of experimental trawling analysed by sediment profile imagery. *J. Exper. Mar. Biol. Ecol.*, **285:** 453-463.

Northeast Region EFHSC (Northeast Region Essential Fish Habitat Steering Committee). 2002. Workshop on the effects of fishing gear on marine habitats off the Northeastern United States October 23-25, 2001 Boston, MA. Northeast Fish. Sci. Cent. Ref. Doc. 02-01. 86 pp.

Pearson, T. H. & Barnett, P. R. O. 1987: Long-term changes in benthic populations in some west European coastal areas. *Estuaries*, **10**, 220-226.

Peterson, C.H., Summerson, H.C. and Fegley, S.R. 1987. Ecological consequences of mechanical harvesting of clams. *Fisheries Bulletin*. **85:** 281–289

Pitcher, C.R., Poiner, I.R., Hill, B.J. & Burridge, C.Y. 2000. Implications of the effects of trawling on sessile megazobenthos on a tropical shelf in north eastern Australia. *ICES. J. Mar. Sci*., 57, 1359-1368.

Prena, J., Schwinghamer, P., Rowell, T.W., Gordon, Jr. D.C., Gilkinson, K.D., Vass, W.P. & McKeown, D.L. 1999. Experimental otter trawling on a sandy bottom ecosystem of the Grand Banks of Newfoundland: analysis of trawl bycatch and effects on epifauna. *Marine Ecological Progress Series.* **181:** 107-124.

Queirós, A.M., Hiddink, J.G., Kaiser, M.J. & Hinz, H. 2006. Effects of chronic bottom trawling disturbance on benthic biomass, production and size spectra in different habitats. *J. Exp. Mar. Biol. Ecol*., **335:** 91-103.

Ragnarsson, S.A. & Lindegarth, M. 2009. Testing hypotheses about temporary and persistent effects of otter trawling on infauna: changes in diversity rather than abundance. *Mar. Ecol. Prog. Ser.,* **385**, 51–64

Rees, H. L. & Eleftheriou, A. 1989. North Sea benthos: A review of field investigations into the biological effect of man's activities. *Journal du Conseil. Conseil international pour l'exploration de la mer*, **45**, 284-305.

Roberts, C., Smith, C., Tillin, H. & Tyler-Walters, H. 2010. Review of existing approaches to evaluate marine habitat vulnerability to commercial fishing activities. Report: SC080016/R3.Environment Agency, Bristol. 150 pp

Sanchez, P., Demestre, M., Ramon, M. & Kaiser, M. J. 2000. The impact of otter trawling on mud communities in the northwestern Mediterranean. *ICES J. Mar. Sci*., **57:** 1352–1358.

Schratzberger, M., Dinmore, T.A. & Jennings, S. 2002. Impacts of trawling on the diversity, biomass and structure of meiofauna assemblage. *Mar. Biol*., **140:** 83-93.

Schwinghamer, P., Gordon, Jr., D.C., Rowell, T.W., Prena, J., McKeown, D.L., Sonnichsen, G. & Guigne, J.Y. 1998. Effects of experimental otter trawling on surficial sediment properties of a sandy-bottom ecosystem of the Grand Banks of Newfoundland. *Cons. Biol*., **12 (6)** 1215-1222.

Schwinghamer, P., Guigne, J.Y. & Siu, W.C. 1996. Quantifying the impact of trawling on benthic habitat structure using high resolution acoustics and chaos theory. *Can. J. Fish. Aquat. Sci*., **53 (2**) 288-296.

Seafish. 2015. Basic fishing methods. A comprehensive guide to commercial fishing methods. August 2015. 104 pp.

Sewell, J. & Hiscock, K. 2005. Effects of fishing within UK European Marine Sites: guidance for nature conservation agencies. Report to the Countryside Council for Wales, English Nature and Scottish Natural Heritage from the Marine Biological Association. Plymouth: Marine Biological Association. CCW Contract FC 73-03-214A. 195 pp.

Shumway, S.E. & Parsons, J.G.J. 2006. Scallops: biology, ecology and aquaculture. *Elsevier Science,* Oxford. **35**

Smith, C.J., Papadopoulou, K.N. & Diliberto, S. 2000. Impact of otter trawling on an eastern Mediterranean commercial trawl fishing ground. *ICES J. Mar. Sci*., **57**: 1340–1351.

Smith, C.R. & Brumsickle, S.J. 1989. The effects of patch size and substrate isolation on colonization modes and rates in an intertidal sediment. *Limnol. Oceanogr*., **3:**, 1263–1277.

Tarnowski, M. 2006. A literature review of the ecological effects of hydraulic escalator dredging. *Fish. Tech. Rep. Ser.* **48**: 30

Thrush, S.F. & Dayton, P.K. 2002. Disturbance to marine benthic habitats by trawling and dredging: implications for marine biodiversity. *Annu. Rev. Ecol. Syst*., **33:** 449-473.

Thrush, S.F., J.E. Hewitt, V.J. Cummings, P.K. Dayton, M. Cryer, S.J. Turner, G.A. Funnell, R.G. Budd, C.J. Milcurn & M.R. Wilkinson. 1998. Disturbance of the marine benthic habitat by commercial fishing: impacts at the scale of the fishery. *Ecol. Appl*., **8(3):** 866-879.

Trimmer, M., Petersen, J., Sivyer, D.B.,Mills, C., Young, E. & Parker, E.R. 2005. Impact of long-term benthic trawl disturbance on sediment sorting and biogeochemistry in the southern North Sea. *Mar. Ecol. Prog. Ser*. **298**: 79–94.

Mayer, L.M., Schick, D.F., Findlay, R.H. and Rice. D.L. 1991. Effects of commercial dragging on sedimentary organic matter. *Marine Environmental Research.* **31:** 249-261.

Tuck, I.D., Hall, S.J., Robertson, M.R., Armstrong, E. & Basford, D.J. 1998. Effects of physical trawling disturbance in a previously unfished sheltered Scottish sea loch. *Mar. Ecol. Progr. Ser*., **162:** 227-42.

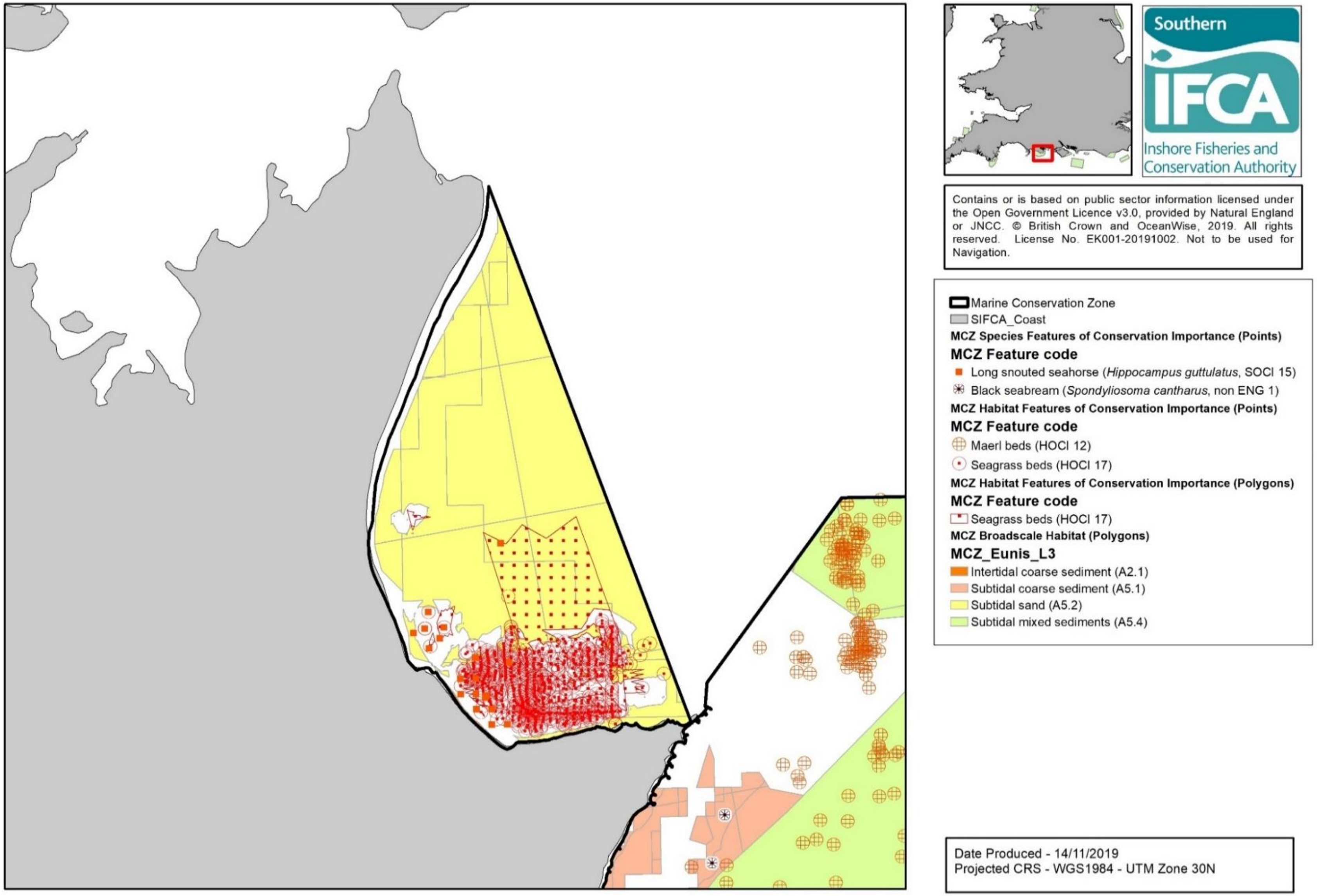
Tudela, 2004. Ecosystem effects of fishing in the Mediterranean: an analysis of the major threats of fishing gear and practices to biodiversity and marine habitats. *Studies and Reviews. General Fisheries Commission for the Mediterranean*. **74:** 44

Valentine, P.C. & Lough, R.G. 1991. The influence of geological and oceanographic environmental factors on the abundance and distribution of fisheries resources of the north eastern United States continental shelf: The seafloor environment and the fishery of eastern Georges Bank. Open File Report 91-439, US Geol. Surv. 25 pp.

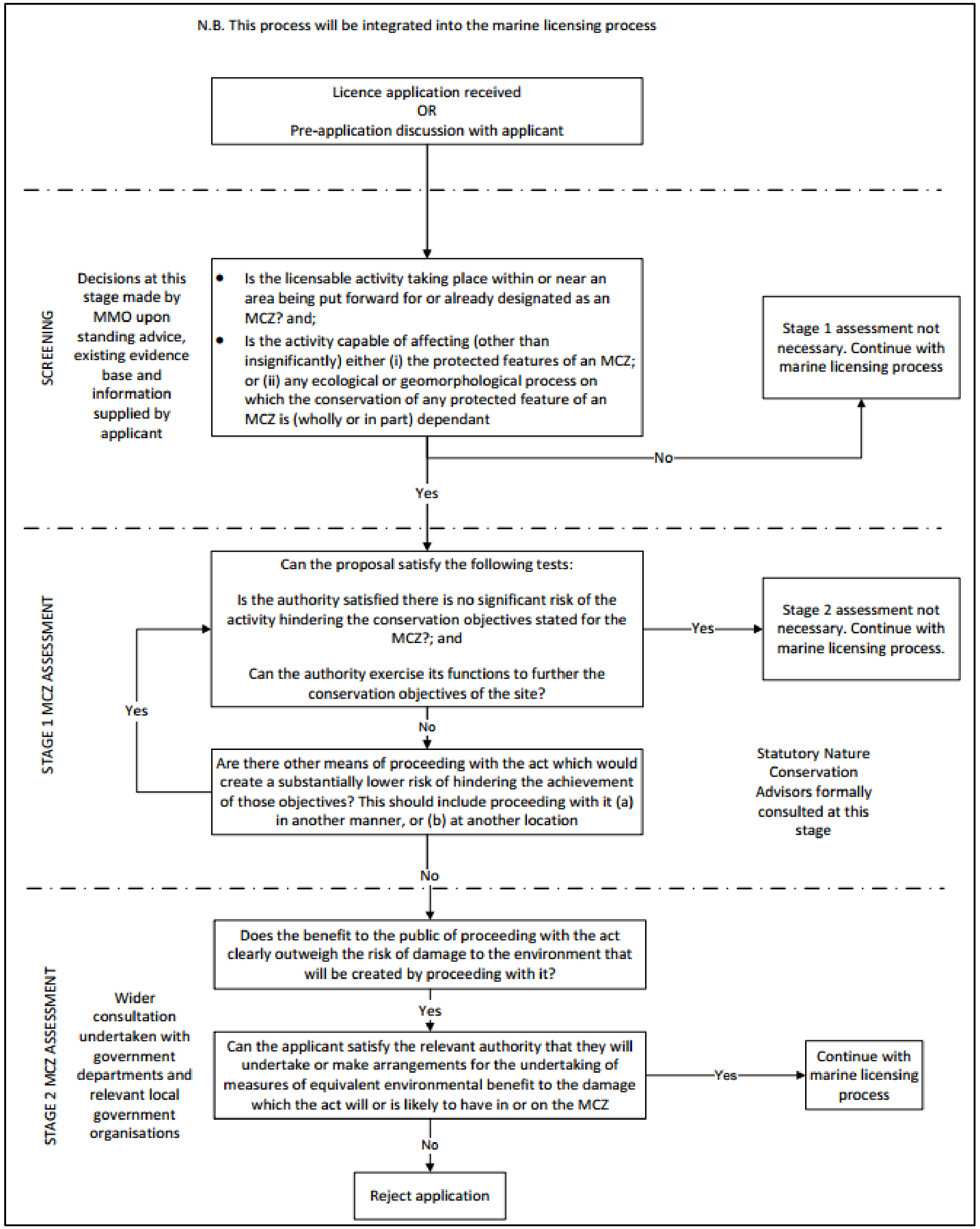
Van Dolah, R. F., Wendt, P. H. & Levisen, M. V., 1991. A study of the effects of shrimp trawling on benthic communities in two South Carolina sounds. *Fish. Res*., **12 (2):** 139-15

Wallace, D.H. & Hoff, T.B. 2005. Hydraulic clam dredge effects on benthic habitat off the north eastern United States. American Fisheries Society Symposium **41**, Bethesda, MD, pp. 691-693.

Annex 1. Broad scale habitat and species features of conservation importance (FOCI) map of the Studland MCZ.



Annex 2. Summary of MMO assessment process for MCZs



Annex 3. Initial screening of commercial fishing activities in the Studland MCZ.

Broad Gear Type (for assessment)	Aggregated Gear Type (EMS Matrix)	Fishing gear type	Does it Occur?	Details	Sources of Information	Potential For Acivity Occur/ Is the activity anticipated to occur?	Justification	Suitable for Part A Assessment?	Priority
Bottom towed fishing gear	**Towed (demersal)**	Beam trawl (whitefish)	N	Currently not known to occur.	Local IFCO	Y	This activity has the potential to occur. Soft bottomed substrate lends itself to this method. One vessel comes into to the district occasionally but it is not known if fish in the MCZ.	Y	Medium to High
		Beam trawl (shrimp)	N		Local IFCO		Target species does not occur.		
		Beam trawl (pulse/wing)	N		Local IFCO		Prohibited via Electric fishing byelaw.		
		Heavy otter trawl	N		Local IFCO	N	The activity has the potential to occur but is not anticipated to occur. The boats which operate within the district (and the Solent) are small in nature (restricted to 12 m or less in length) and so are restricted in the size of gear used. This means light otter trawls are used instead of heavy otter trawls.		
		Multi-rig trawls	N		Local IFCO	N	Has not historically occurred and is not currently anticipated to occur, the boats which operate within the district (and the Solent) are small in nature (restricted to 12 m or less in length) and so are restricted in the size of gear used. This means multi rig trawls are not used and the activity is not anticipated to occur.		

		Light otter trawl	N		Local IFCO	Y	The activity has the potential to occur and the target species is likely to occur. However, there are currently no vessels actively trawling in this area.	Y	High
		Pair trawl	N		Local IFCO	N	It is not anticipated to occur as it has not historically occurred. Furthermore, there is limited potential due to the space required to accommodate two vessels and the size/power of vessels needed.		
		Anchor seine	N		Local IFCO	N	Gear type has not been historically used within the area and is not anticipated to occur. Activity needs a large area and, in the site, considered would be limited. In addition, large vessels are also required for this gear type and vessels over 12 m in length are prohibited from fishing within the Southern IFCA district.		
		Scottish/fly seine	N		Local IFCO	N	Gear type has not been historically used within the area and is not anticipated to occur. Activity needs a large area and, in the site, considered would be limited. In addition, large vessels are also required for this gear type and vessels over 12 m in length are prohibited from fishing within the Southern IFCA district.		
Pelagic towed fishing gear	**Towed (pelagic)**	Mid-water trawl (single)	N		Local IFCO	N	Gear type has not been historically used within the area. Activity has the potential to occur however this gear type does not come into contact with the seabed and therefore there is no chance for interaction with designated features.		

		Mid-water trawl (pair)	N		Local IFCO	N	Gear type has not been historically used within the area. Furthermore, there is limited potential due to the space required to accommodate two vessels and the size/power of vessels needed. This gear type does not come into contact with the seabed and therefore there is no chance for interaction with designated features.		
		Industrial trawls	N		Local IFCO	N	Activity is not able to occur due to the size of vessel required. Vessels over 12 m are prohibited from fishing within the Southern IFCA district.		
Bottom towed fishing gear	**Dredges (towed)**	Scallops	N		Local IFCO	N	Gear type has not historically occurred within the site and is not anticipated to occur.		
		Mussels, clams, oysters	N		Local IFCO	N	Historic trails of this method of fishing in the area proved not commercially viable due to species present and substrate type. Therefore, it is not anticipated that the activity will occur.	Y	Medium to High
		Pump scoop (cockles, clams)	N		Local IFCO	N	Historic trails of this method of fishing in the area proved not commercially viable due to species present and substrate type. Therefore, it is not anticipated that the activity will occur.		Medium to High
Suction	**Dredges (other)**	Suction (cockles...)	N	Not allowed in the district.	Local IFCO	N	Suction dredging for cockles, clams, mussels and oysters is prohibited (by default) in the Southern IFCA district (by Southern IFCA byelaws).		
Tractor		Tractor	N		Local IFCO	N	The activity has not historically occurred within the site and is not anticipated to occur.		

Intertidal work	**Intertidal handwork**	Hand working (access from vessel)	N		Local IFCO	Y	Hand working with access from a vessel infers a muddy habitat where there difficulty accessing areas. At this site, the dominance of sand and coarse sediment substrate means there is limited need for a vessel as the substrate means the area is accessible on foot.		
		Hand work (access from land)	Y		Local IFCO	Y	The activity is known to occur within the site.	Y	low to medium
Static - pots/traps	**Static - pots/traps**	Pots/creels (crustacea/gastropods)	Y		Local IFCO	Y	Activity is known to occur. In the Area but not inside the MCZ.	Y	low
		Cuttle pots	Y	Unknown	Local IFCO	Y	Activity is known to occur.		low
		Fish traps	N		Local IFCO	N	Activity has not historically occurred within the site and is not anticipated to occur.		
Demersal nets/lines	**Static - fixed nets**	Gill nets	Y	Up to six vessels may net in the MCZ. Targeting plaice, sole, ray skate.	Local IFCO		It is anticipated that static fixed nets are used within the site in areas of shallow water, although effort is likely to be low with the area worked by 1 to 2 vessels at a time. The activity is unlikely in deeper water due to the rushing tide in the outer reaches of the site.	Y	Low to Medium
		Trammels	Y	See 'gill nets'	Local IFCO		See 'gill nets'		Low to Medium
		Entangling	Y	See 'gill nets'	Local IFCO		See 'gill nets'		Low to Medium

Pelagic nets/lines	**Passive - nets**	Drift nets (pelagic)	N		Local IFCO	N	Activity is not anticipated to occur and potential for the activity is limited by shallow waters and the rushing tide that effects the site, particularly the outer areas.		
Demersal nets/lines		Drift nets (demersal)	Y		Local IFCO	Y		Y	low to medium
	Lines	Longlines (demersal)	Y		Local IFCO	Y	It is anticipated that demersal longlines are used within the site,	Y	
Pelagic nets/lines		Longlines (pelagic)	N		Local IFCO	N	The activity has not historically occurred within the site and is not anticipated to occur.		
		Handlines (rod/gurdy etc)	Y	The activity is known to occur however this gear type does not come into contact with the seabed and therefore there is no chance for interaction with designated features. Shore-based angling is limited and due to the nature of the shoreline is unlikely to interact with venerable	Local IFCO		The activity is known to occur however this gear type does not come into contact with the seabed and therefore there is no chance for interaction with designated features. Shore-based angling is limited and due to the nature of the shoreline is unlikely to interact with venerable designated features.		

				designated features.					
		Jigging/trolling	Y	See 'handlines (rod/gurdy etc)'	Local IFCO		See 'handlines (rod/gurdy etc)'		
Purse seine	**Seine nets and other**	Purse seine	N		Local IFCO	N	Activity has not historically occurred within the site and is not anticipated to occur.		
Demersal nets/lines		Beach seines/ring nets	N		Local IFCO	Y	The activity has not historically occurred within the site but has the potential to occur. Possible ring netting for mullet maximum 6 vessels.	Y	

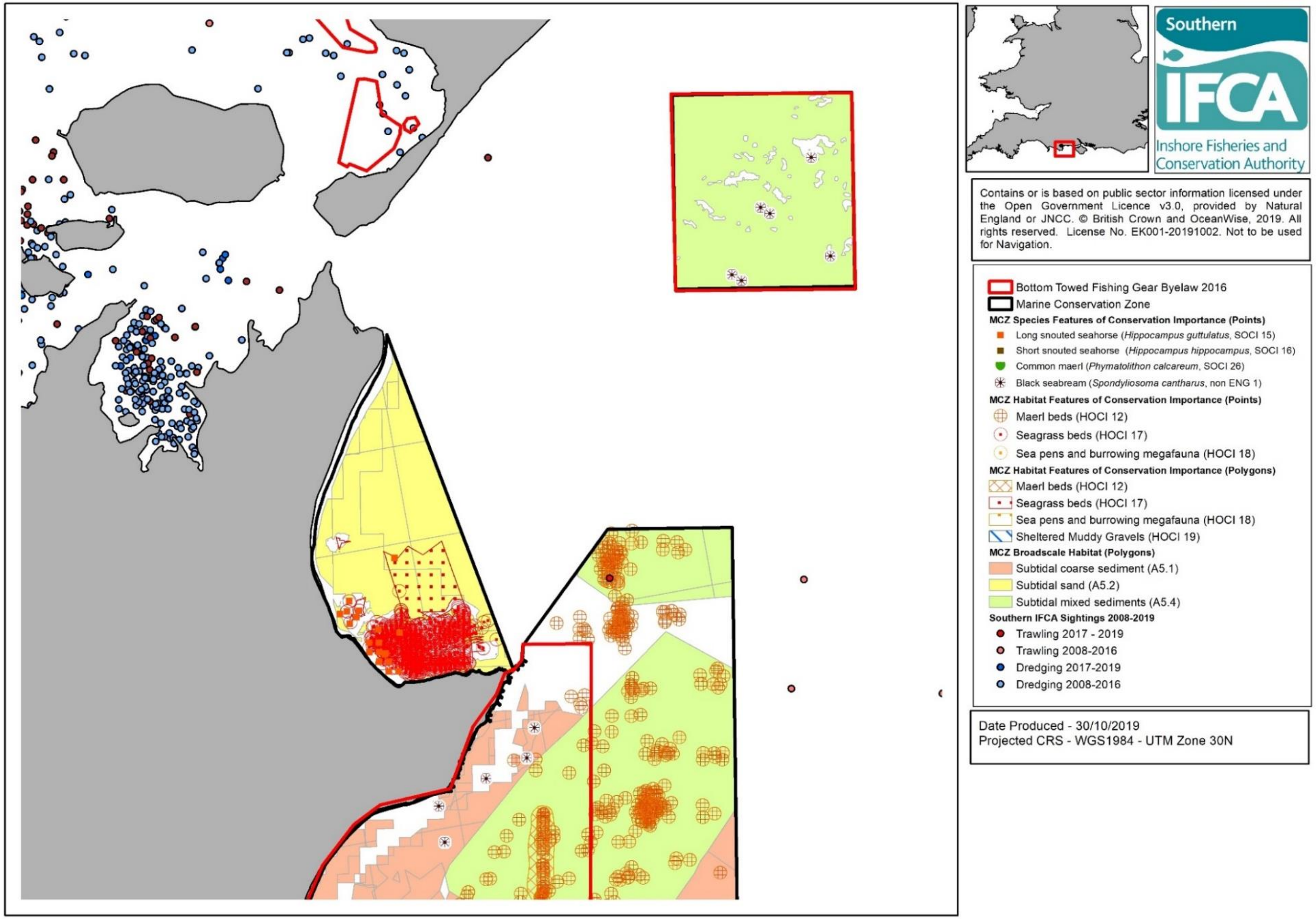
Miscellaneous		Shrimp push-nets	N	Unknown	Local IFCO		The occurrence of the activity is unknown. It is not anticipated to occur as it is not thought to have occurred historically within the site. The activity has the potential to occur but is unlikely to because of a lack of areas with suitable substrate to support the target species. In addition, activity is conducted intertidally and designated features are not intertidal and therefore whilst there is limited potential for the activity to occur it will not take place over designated features.		
EA Only		Fyke and stake nets			EA Only			EA Only	
Miscellaneous	**Miscellaneous**	Commercial diving	N			N	Activity has not historically occurred and is not anticipated to occur.		
Bottom towed fishing gear		Bait dragging	N			N	Activity has not historically occurred within the site and is not anticipated to occur. The majority substrate present is not suitable for the activity to take place. As such, the target species are also not present.		
Miscellaneous		Crab tiling	N			N	Activity has not historically occurred within the site or Southern IFCA district and therefore is not anticipated to occur.		
Intertidal work	**Bait collection**	Digging with forks	Unknown			Y	Activity has the potential to occur as the site may support lugworm, and access to the intertidal is possible by foot.	Y	

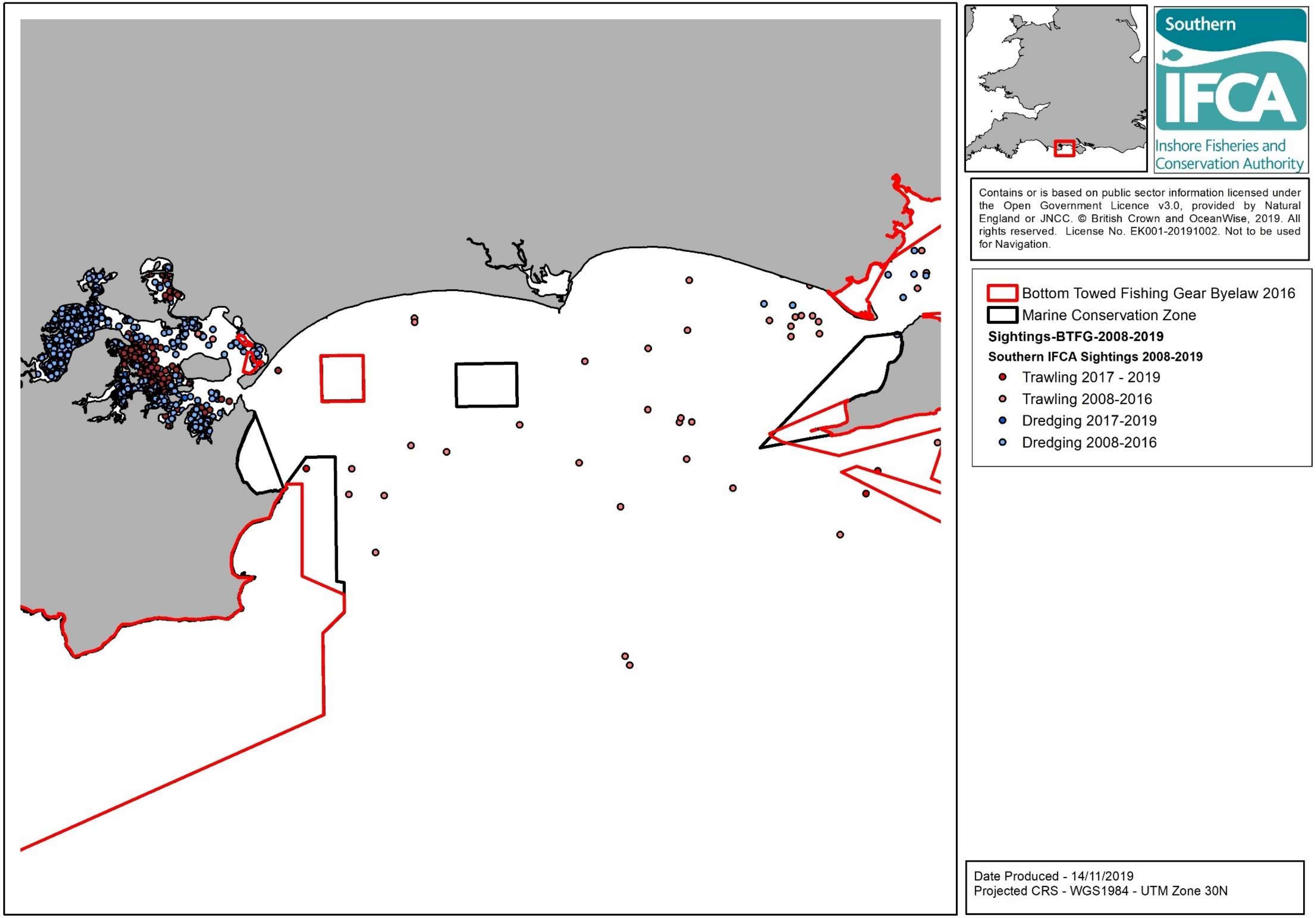
Annex 4. Natural England's Advice on Operations for Studland Bay MCZ (a) dredging (b) trawling.

Pressure Name	Habitat			Species
	Intertidal coarse sediment	Seagrass beds	Subtidal sand	Long snouted seahorse
Abrasion/disturbance of the substrate on the surface of the seabed	NS	S	S	IE
Changes in suspended solids (water clarity)	NS	S	S	S
Penetration and/or disturbance of the substratum below the surface of the seabed, including abrasion	NS	S	S	
Removal of non-target species		S	S	S
Smothering and siltation rate changes (Light)	NS	S	S	
Collision BELOW water with static or moving objects not naturally found in the marine environment				IE
Deoxygenation	NS	NS	S	S
Hydrocarbon & PAH contamination	NA	NA	NA	NA
Introduction of light		S	S	
Introduction or spread of invasive non-indigenous species (INIS)		S	S	IE
Litter	NA	NA	NA	IE
Nutrient enrichment	NS	S	NS	
Organic enrichment	NS	S	S	
Physical change (to another sediment type)	S	S	S	
Synthetic compound contamination (incl. pesticides, antifoulants, pharmaceuticals)	NA	NA	NA	NA
Transition elements & organo-metal (e.g. TBT) contamination	NA	NA	NA	NA
Underwater noise changes			NS	S
Visual disturbance			NS	S

Pressure Name	Habitat			Species
	Intertidal coarse sediment	**Seagrass beds**	**Subtidal sand**	**Long snouted seahorse**
Abrasion/disturbance of the substrate on the surface of the seabed	NS	S	S	IE
Changes in suspended solids (water clarity)	NS	S	S	S
Penetration and/or disturbance of the substratum below the surface of the seabed, including abrasion	NS	S	S	
Removal of non-target species		S	S	S
Removal of target species		S	S	
Smothering and siltation rate changes (Light)	NS	S	S	
Visual disturbance			NS	S
Collision BELOW water with static or moving objects not naturally found in the marine environment				IE
Deoxygenation	NS	NS	S	S
Hydrocarbon & PAH contamination	NA	NA	NA	NA
Introduction of light		S	S	
Introduction of microbial pathogens		S	S	S
Introduction or spread of invasive non-indigenous species (INIS)		S	S	IE
Litter	NA	NA	NA	IE
Nutrient enrichment	NS	S	NS	
Organic enrichment	NS	S	S	
Physical change (to another sediment type)	S	S	S	
Synthetic compound contamination (incl. pesticides, antifoulants, pharmaceuticals)	NA	NA	NA	NA
Transition elements & organo-metal (e.g. TBT) contamination	NA	NA	NA	NA
Underwater noise changes			NS	S

Annex 5. Fishing activity maps using trawl and dredge sighting data from 2008-2019 in (a) Studland MCZ and (b) Poole Bay.




Southern
IFCA
Inshore Fisheries and Conservation Authority
Bottom Towed Fishing Gear Byelaw 2016
Marine Conservation Zone
Sightings-BTFG-2008-2019
Southern IFCA Sightings 2008-2019
Trawling 2017 - 2019
Trawling 2008-2016
Dredging 2017-2019
Dredging 2008-2016
Date Produced - 14/11/2019
Projected CRS - WGS1984 - UTM Zone 30N