

Fisheries Impacts on Marine Protected Habitats – A Review of the Evidence

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Further information

This report can be downloaded from the [Natural England Access to Evidence Catalogue](#). For information on Natural England publications contact the Natural England Enquiry Service on 0300 060 3900 or e-mail enquiries@naturalengland.org.uk.

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Executive summary

This report contains several literature synthesis regarding the impacts of different fishing gears on a variety of marine habitats that are protected within English waters. These syntheses were produced by Natural England as part of a project commissioned by DEFRA's Marine Biodiversity Impacts Evidence Group. The purpose of the report is to support the assessment and management of commercial fishing in Marine Protected Areas (MPAs), providing a summary of evidence to inform Habitat Regulations and Marine Conservation Zone Assessments.

Terminology and definitions used within the reviews is standardised to ensure consistency between this report and Natural England's formal published MPA Conservation Advice packages. This standardisation therefore covers the categorisation of habitats and fishing gears, as well as structuring the reviews in accordance with the key pressures that those gears exert on marine habitats. This standardisation should allow for easier incorporation of evidence into MPA assessments. Further guidance on MPA conservation advice and use within assessments can be found via [Guidance on how to use Natural England's Conservation Advice Packages for Environmental Assessments](#).

Natural England has endeavoured to use the best available evidence within this report and where possible, the report also highlights key evidence gaps and limitations. Ultimately, the scale and significance of impacts are very dependent upon the exact gear and that gear's interaction with the habitat. Variables such as footprint of interaction, longevity of interaction and habitat recovery times are important factors in determining overall impact. Therefore, whilst the report should aid MPA assessments, site specific evidence and advice must also be used to inform HRA's and any subsequent fisheries management decisions.

It should be noted that a further aspect of the project commissioned by the Impacts Evidence Group was to create a database that records the available evidence on fisheries impacts on MPA features. This database will be published and made available to the public soon.

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1. Scallop dredging on subtidal mud, subtidal sand, subtidal coarse sediment, and subtidal mixed sediment

1.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports. Most information has come from studies undertaken in UK waters, though a few studies from elsewhere have also been included where their evidence is relevant.

1.2. Background

Scallop dredging is one of the most damaging forms of fishing to benthic habitats, although these effects vary among different habitats (Szostek *et al.*, 2016). In a mixed sand, gravel, and cobble habitat, Sciberras *et al.* (2013) found that short-term effects of scallop dredging could not be differentiated from effects caused by natural disturbance (strong currents, tides, or storm events). However, chronic fishing pressure that is greater in frequency and/or magnitude than natural disturbance, can alter community structure and function, remove biomass, and reduce production (Hiddink *et al.*, 2006). This summary reviews the impacts of gears used to dredge for both king scallop *Pecten maximus* and queen scallop *Aequipecten opercularis* in UK waters.

1.3. Impacts

1.3.1. Abrasion and penetration (to the physical habitat and damage to epifauna and infauna)

The effects of scallop dredging on marine ecosystems vary with different seabed types, levels of background disturbance, local hydrography, fishing intensity and the characteristics of the ecological community associated with the habitat in question (Bradshaw *et al.*, 2001). In general, scallop dredging causes loss of biodiversity and reduces the complexity of benthic habitats by flattening substrates and removing structurally complex species such as hydroids, bryozoans, and seaweeds (Sewell & Hiscock, 2005). Such habitats are key nursery and feeding areas for a wide range of species, including commercially important fish and shellfish (Stewart & Howarth, 2016). Overall, species diversity and richness, the total number of species and the number of individuals, are found to decrease significantly with increased fishing effort (Veale *et al.*, 2000a).

The magnitude of impact varies considerably depending upon the habitat in question. More mobile sediments, such as sand, mud and to a lesser extent gravel, appear more resilient than others, particularly in areas adapted to high levels of natural disturbance (Stewart & Howarth, 2016). Subtidal sand habitats, subject to natural periodic wave or current disturbance, have often been found to be little affected and/or to recover within a few weeks after single passes (Sewell & Hiscock, 2005). Determining the full effects of dredging remains difficult, as most fishing grounds have been exploited for decades, long before scientific study began (Stewart & Howarth, 2016).

Due to their penetrative nature and close contact with the seabed, scallop dredges cause substantial physical disruption to the seafloor by ploughing sediments and damaging organisms attached to or resting upon the seabed (epifauna), such as hydroids, bryozoans, and sponges (Kaiser *et al.*, 2000). In addition, the disturbance of a passing dredge can negatively affect the ability of scallops to swim and escape predators (Maguire *et al.*, 2002).

Jenkins *et al.* (2001) found the species most sensitive to dredge damage around the Isle of Man to be the seven-armed starfish *Luidia ciliaris*, the sea urchin *Echinus esculentus* and the brown crab *Cancer pagurus*; and in the Firth of Lorn, bycatch species (in addition to those above) included the common starfish *Asterias rubens* and the bivalves *Neptunea antiqua* and *Glycymeris* (Boulcott *et al.*, 2014). Initial contact with the dredge teeth appears to cause most of the fatal damage (Jenkins *et al.*, 2001), while non-fatal damage appears to occur in the mesh bag during the tow and landing of the catch (Shephard *et al.*, 2009). Jenkins *et al.* (2001) found that over 75% of the megafauna which encountered scallop dredges remained in the seafloor, having been crushed when passed around, through or under the heavy gear or by initial encounter with the tooth bar.

Typically, stable mixed sediment seabeds (cobble, pebble, gravel, shell debris, sand, and mud mixtures) are dominated by faunal turfs consisting largely of erect hydroids (e.g., *Nemertesia* spp., *Obelia* spp., *Abietinaria abietina*) and erect bryozoans (e.g., *Flustra foliacea*, *Bugula* spp., *Alcyonidium diaphanum*), all of which are particularly vulnerable to scallop dredging (Stewart & Howarth, 2016; Bradshaw *et al.*, 2000; Eleftheriou & Robertson, 1992). These species form emergent structures that provide important settlement substrates for many other species, including scallop spat (Lambert *et al.*, 2011; Bradshaw *et al.*, 2001). The abundance of species within such faunal turfs has been found to be reduced by 56-96% by dredging (Kaiser *et al.*, 2006).

As scallop dredges can penetrate 3-10 cm into the seabed, they have a strong potential to disrupt benthic infauna (Stewart & Howarth, 2016). However, consistent proof of this has been harder to determine: some studies have found little change in the abundance and biodiversity between dredged and undredged sites (e.g., O'Neill *et al.*, 2013), whereas others report a significant reduction in infaunal biomass (e.g., Kaiser *et al.*, 2000). In this latter study, communities within areas closed to scallop dredging were dominated by higher biomass and emergent fauna, both of which increased habitat complexity. By contrast, areas fished by towed gear were dominated by smaller-bodied fauna and scavenging taxa (Kaiser *et al.*, 2000). Large infaunal species, such as the burrowing heart urchin *Echinocardium cordatum*, razor shells *Ensis* spp. and burrowing sand eels

Ammodytes, are frequently destroyed by dredging operations (Eleftheriou & Robertson, 1992). Pranovi *et al.* (2000) also found adverse effects to infaunal polychaetes and amphipods.

1.3.2. Changes in suspended solids; smothering and siltation

Scallop dredges are responsible for mobilising sediment, with an associated release of nutrients and possible contaminants (O'Neill *et al.*, 2013). Those sediments subject to high levels of natural disturbance (i.e., the finer fractions influenced to a greater degree by currents and waves) appear to be more resilient to disturbance from dredging (Stewart & Howarth, 2016). However, where these sediments lie close to rocky reefs, settling sand particles can adversely impact sessile epifaunal communities associated with the reefs (Dale *et al.*, 2011); and deposition of finer suspended sediments can lead to smothering of feeding and respiratory organs of some sessile epifaunal species (Jones, 1992; Kaiser *et al.*, 2002).

1.3.3. Removal of species

In addition to capturing scallops, dredges capture a wide variety of non-target mobile megafauna. Examples include fishes (flatfish, dogfish, skates, rays, monkfish, and dragonets), crustaceans (edible crabs, swimming crabs, spider crabs and hermit crabs), echinoderms (brittlestars, starfish and sea urchins), and molluscs (bivalves, gastropods, and cephalopods) (Stewart & Howarth, 2016). In a study undertaken off the Isle of Man, Hinz *et al.* (2012) found that for every scallop captured by a Newhaven dredge, four non-target individuals were also caught. Szostek *et al.* (2017) determined that bycatch biomass (as a proportion of total catch biomass) varied between Cardigan Bay off west Wales (15%), the English Channel (19%) and around the Isle of Man (53%). Öndes *et al.* (2016) found that up to 4.8% of brown crab *Cancer pagurus* landings around the Isle of Man were affected by scallop dredging bycatch mortalities.

On mixed substrata in particular, species composition in dredged areas has been found to differ greatly compared to undredged areas (Sewell & Hiscock, 2005). Scallop dredging may significantly reduce the number of species, number of individuals and lower biomass of macrofauna (Pranovi *et al.*, 2000). Indeed, the greatest amount of mortality is left on the seabed rather than occurring as bycatch (Jenkins *et al.*, 2001).

Following dredging, predatory mobile species such as fish, crabs and starfish have been found to be attracted to the dredge tracks within one hour of dredging at densities up to 30 times greater than in areas outside the tracks (Maguire *et al.*, 2002). The supplementation of the diet of predators such as starfish or crabs from carrion left in the dredge tracks (as noted by Veale *et al.*, 2000b) may lead to shifts in benthic community structure.

1.3.4. Recovery of habitats and communities

In general, the more naturally stable an area of seabed is, the more sensitive the ecological community appears to be to disturbance (Stewart & Howarth, 2016). It has

therefore been suggested that the effects of dredging will be relatively short-lived for ecological communities adapted to frequent natural disturbance by currents, tides, storms, and re-suspension of sediment, such as those inhabiting soft mud/sand/sandy gravel sediments (Jennings & Kaiser, 1998; Sciberras *et al.*, 2013). Slow-growing species (such as sponges or soft corals) take much longer to recover (up to 8 years) from scallop dredging than organisms with shorter lifespans such as polychaete worms and encrusting bryozoans (less than one year) (Kaiser *et al.*, 2006; Hinz *et al.*, 2011). This can lead to the benthic community shifting from one dominated by upright species to one dominated by small, encrusting, opportunistic, fast-growing species (Bradshaw *et al.*, 2001). Over a 60-year period, Bradshaw *et al.* (2002) found that the amount of change in the benthic community was related to how long a site had been fished, rather than actual fishing intensity. Mobile, robust, and scavenging taxa had increased in abundance, while slow-moving or sessile, fragile taxa had decreased.

2. Beam trawling (for whitefish) on subtidal sand

2.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports, the majority of which relate to UK waters. It is based upon an earlier draft prepared by staff at the Marine Management Organisation.

2.2. Background

This review assesses the impacts when a heavy beam trawl (where the mouth of the net is held open by a steel or wooden pole, typically 4 m long on UK vessels but which may be up to 12 m long on some continental vessels) targeting demersal fishes (typically flatfishes, especially sole and plaice, but also cod, whiting and haddock) is towed over a seabed consisting primarily of sand, periodically affected by tidal currents and/or wave action. In UK fleets, two trawls are often towed from both sides of the one vessel at the same time. In deeper waters, megrim sole and monkfish may also be targeted (Seafish, 2021).

2.3. Impacts

2.3.1. Abrasion and penetration (to the physical habitat and damage to epifauna and infauna)

Contact with any part of a beam trawl (beam, shoes, mats, nets, and chains) will have a direct effect on the seabed. Jones (1992) described these effects as being scraping and

ploughing (now referred to as abrasion and penetration); sediment re-suspension (smothering and siltation rate changes); and removal or scattering of non-target organisms. By penetrating a sandy seabed, a beam trawl will leave behind track marks and trenches, and alter the bathymetry (Tuck et al., 1998; Grieve et al., 2015; Gravestock, 2017). Of all the beam trawl gear components, the tickler chains, positioned in front of the ground rope, are reported to cause the deepest penetration in sandy bottom habitats (Sewell and Hiscock, 2005; Depestele et al., 2015; Grieve et al., 2015; Szostek et al., 2017). Penetration depths range from a few mm to 6 cm (Bergman & Hup, 1992; Paschen et al., 2000; Depestele et al., 2015; Eigaard et al., 2016; Szostek et al., 2017). The depth difference of sandy seabed bathymetry after multiple passes of a beam trawl ranges between 0 and 127.9 mm and can be 28.5 mm after a single pass of a beam trawl (Depestele et al., 2015).

Sediment compaction of sand, causing hardness, can result from passage of beam trawls (Depestele et al., 2015; Johnson et al., 2017); and it has been found that beam trawl tracks are considerably more noticeable on muddy sand than on hard, coarse sand (Margetts & Bridger, 1971; de Groot, 1984).

Chronic beam trawling (on average 6.5 times per year) in an area of the North Sea characterised by muddy sand at 55-75 m depth led to significant declines in infaunal productivity and biomass (Jennings et al., 2001; Kaiser, 2014). However, less intensive beam trawling (on average 2.3 times per year) in an area of the North Sea characterised by sand at 40-65 m depth had no significant effect on infauna, suggesting that beam trawling at frequencies of less than three times per year does not have adverse, long-term effects on benthic communities in sandy habitats (Jennings et al., 2001; Kaiser, 2014). Kaiser et al. (2006) also found negative impacts of beam trawling on benthic taxa in muddy-sand and sand habitats to be short-term.

High proportions of long-lived taxa are found in subtidal sand and macrofaunal species in sandy and silty areas have generally shown high mortalities in response to beam trawling, in comparison to smaller, short-lived species (Bergman & Van Santbrink, 2000; Løkkeborg, 2005; Rijnsdorp et al., 2016). Beam trawling on stable sand and gravelly areas was found to lead to declines in abundance of up to 50% for nine of the most common taxa (Kaiser & Spencer, 1996; Løkkeborg, 2005).

Following a beam trawl pass, damaged animals rapidly attract scavengers (Sewell & Hiscock, 2005). Highly mobile scavengers, such as fish and crabs, quickly arrive at beam trawl tracks within minutes to hours, dispersing once feeding has taken place (Kaiser & Spencer, 1996). Whelks (*Buccinum undatum*) have been shown to survive beam trawling and are capable of exploiting a wide variety of prey, feeding on damaged and moribund animals in trawled areas (Evans et al., 1996). Fish such as gurnard, whiting and dogfish, and the sea urchin *Strongylocentrotus pallidus*, are also known to aggregate over beam trawl tracks to feed (Kaiser & Spencer, 1994).

2.3.2. Changes in suspended solids; smothering and siltation

A sediment plume is generated by the passage of the trawl, releasing nutrients and possible pollutants into the water column and increasing the total suspended sediment load (Jones, 1992). Clean sandy sediments are likely to produce less suspended sediment than say, muddy sand, due to the lower proportion of silt present within the surface layers.

2.3.3. Removal of species

Beam trawling catches a large range of bottom living species and is not a well-targeted fishery, often with poor selectivity and the potential to catch a wide variety of non-target bycatch (Seafish.org - accessed 15Jan21). Beam trawling has negative effects on non-target species and benthic communities, resulting in declines in productivity and biomass and high mortality rates recorded for various benthic organisms (Bergman & Hup, 1992; Lokkeborg, 2005; Sewell et al., 2007; Gravestock, 2017).

Changes in benthic community structure are known to occur following beam trawling but the effects can be variable (Lindeboom & de Groot, 1998; Jennings & Kaiser, 1998). Fragile infaunal species which live on or within the surface sediments (such as bivalves, holothurians, and gastropods) are particularly vulnerable to damage or disturbance (Kaiser & Spencer, 1996). Bergman and Van Santbrink (2000) found mortalities up to 52% for echinoderms, up to 39% for crustaceans and gastropods and up to 64% of bivalves after a single beam trawl sweep, although mortalities were lower with trawls using chain mats compared to those using tickler chains. Bergman & Hup (1992) found significant declines (40-65%) in starfish, small crustaceans, heart urchins, and tube-dwelling polychaete worms after beam trawling; the impact appeared to be greatest on concentrations of small-sized individuals, possibly because larger animals live deeper in the sediment or have better escape possibilities. In their global review, Kaiser et al. (2006) found that, in sand, the initial impact of beam trawls on crustaceans, molluscs and echinoderms was more adverse than that on annelids. Mortality from beam trawling may be related to hydrodynamic conditions and species' ability to withstand disturbance (Kaiser & Spencer, 1996).

2.3.4. Recovery of habitats and communities

The recovery time of benthic biota in sandy habitats to beam trawling is variable, with studies reporting biota abundance recovering within 50 to 236 days of a trawling event (Kaiser *et al.*, 1998, 2006; Foden *et al.*, 2010). Kaiser *et al.* (1998) assessed changes which had taken place to megafaunal benthic communities from two different habitats (one with stable sediments and a rich fauna; the other with mobile sediment and a relatively impoverished fauna), six months after beam trawling had taken place. Immediately after fishing, the stable sediment community was significantly altered: the abundance of some species had decreased (e.g., the sea mouse *Aphrodita aculeata*), while others had apparently increased (e.g., the hermit crab *Pagurus bernhardus*), although there was considerable variation between samples. This suggested the effects of trawling were not uniform. For mobile sediment, no effects of trawling were apparent, whereas on more

stable sediments, it could take up to six months before all signs of trawling disturbance had disappeared.

Sewell & Hiscock (2005) pointed out that areas which have been intensively trawled for several years still support profitable fisheries which would not be possible without ample benthic food. It has therefore been suggested that it seems not unlikely that the benthic community in these areas has shifted towards a dominance of highly productive, opportunistic species such as polychaetes (Rijnsdorp *et al.*, 1998; Jennings & Kaiser, 1998).

Meiofauna (organisms between 0.05-0.5 mm) are found to be fairly resistant to beam trawling, more so than macrofauna, with the ability to withstand the effects of intense beam trawling while continuing to carry out essential energy cycling processes (Schratzberger *et al.*, 2002).

3. Beam trawling (for whitefish) on subtidal coarse sediment and subtidal mixed sediment

3.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports, the majority of which relate to UK waters. During the review of evidence, it has not always been possible to identify the precise gear being used, or the sediment type the gear is being dragged over, to be able to characterise the impact that results.

3.2. Background

This review assesses the impacts when a heavy beam trawl (where the mouth of the net is held open by a steel or wooden pole, typically 4 m long on UK vessels but which may be up to 12 m long on some continental vessels), targeting demersal fishes (typically flatfishes, especially sole and plaice, and also cod, whiting and haddock), is towed over a seabed consisting of mud, coarse sand, gravel, pebbles and cobbles periodically affected by tidal currents and/or wave action. In UK fleets, two trawls are often towed from both sides of the one vessel at the same time. Note that in deeper waters, megrim sole and monkfish may also be targeted (Seafish, 2021).

3.3. Impacts

3.3.1. Abrasion and penetration (to the physical habitat and damage to epifauna and infauna)

Beam trawling causes physical disruption of the seabed through contact of the gear components with the sediment and the resuspension of sediment into the water column in the turbulent wake of the gear (Depestele *et al.*, 2015). Contact with any part of the beam trawl (beam, shoes, mats, nets, and chains) will have a direct effect on the seabed. Jones (1992) described these effects can be classified as scraping and ploughing (now referred to as abrasion and penetration); sediment re-suspension (smothering and siltation rate changes); and removal or scattering of non-target organisms. The extent to which the seabed is affected depends on the actual type of fishing gear, the substratum, and its physical characteristics (De Groot & Lindeboom, 1994; Lindeboom & de Groot, 1998; Jennings & Kaiser, 1998; Auster, 1998). A study of beam trawl fisheries within the North Sea by Depestele *et al.* (2015) found that trawl gear caused abrasion to the seabed and could penetrate at least the top 6 cm of sediment, largely through the action of the heavy tickler chains positioned in front of the groundrope. Such disturbance can lead to a direct release of nutrients which can enhance primary production in the water column (Couceiro

et al., 2013) and may release contaminated material if present (Bradshaw *et al.*, 2012). It can also lead to immediate declines in benthic community metabolism (Tiano *et al.*, 2019).

Changes in benthic community structure are known to occur following beam trawling but the effects can be variable (Lindeboom & de Groot, 1998; Jennings & Kaiser, 1998). Fragile infaunal species which live on or within the surface sediments (such as bivalves, holothurians, gastropods) are particularly vulnerable to damage or disturbance (Kaiser & Spencer, 1996). Bergman & Hup (1992) examined the effects of three passages of a trawl over two days, recording a significant lowering of densities (by 40-60%) of echinoderms such as common starfish *Asterias rubens* and sea potato *Echinocardium cordatum*, and of polychaete worms such as sandmason *Lanice conchilega*. The impact appeared to be greatest on densities of small individuals, possibly because larger animals live deeper in the sediment or have better escape possibilities. Other studies, however, found some increases in polychaete numbers may occur following beam trawling, such as *Magelona papillicornis* (Bergman & Hup, 1992), *Chaetozone setosa* and *Caulleriella zetlandica* (Tuck *et al.*, 1998).

Regarding epifauna, beam trawl ground gear (the shoes, tickler chains or chain mat) is known to crush or dislodge animals on the seabed (Revill & Jennings, 2005). Declines in suspension feeders (de Juan *et al.*, 2007), epifauna (Buhl-Mortensen *et al.*, 2016), annual faunal production (Hermesen *et al.*, 2003), biomass, species richness, species diversity, and habitat complexity (Collie *et al.*, 1997; Thrush & Dayton, 2002) have also been attributed to beam trawling.

3.3.2 Changes in suspended solids; smothering and siltation

A sediment plume generated by the passage of the trawl will reduce light levels reaching the substrate, release nutrients and possible pollutants into the water column, and increase the total suspended sediment load (Jones, 1992). Subsequent deposition of the suspended sediment can also result in the smothering of feeding and respiratory organs of sessile organisms, potentially affecting biota in a wide area as the sediment is transported in the water column (Jones, 1992; Kaiser *et al.*, 2002). The seabed type will determine the amount of fine material being re-suspended: those with higher mud fractions will generate more than those which are naturally 'cleaner'.

The upper layers of marine sediments act as an important site for carbon storage (Luisetti *et al.*, 2019) also in the process of nitrogen cycling (van de Velde *et al.*, 2018). Disturbance of these layers will disrupt such processes significantly (van de Velde *et al.*, 2018).

3.3.3 Removal of species

Beam trawling catches a large range of bottom living species and is not a well-targeted fishery with often poor selectivity and the potential to catch a wide variety of non-target bycatch (www.seafish.org - accessed 15Jan21). Kaiser & Spencer (1995) found that starfishes, hermit crabs and molluscs were highly resistant to capture by a 4 m beam trawl, whereas fishes (except dogfish), sea urchins and swimming crabs suffered higher mortality

after capture. Generally, many of the animals that passed through the meshes of the cod end, survived. The chain matrix fitted to the gear was largely responsible for the injuries sustained. Animals that are caught in the trawl itself were highly likely to die, either during hauling or when the catch is being processed and animals are being discarded (Revill & Jennings, 2005).

Beam trawls tend to catch much more bycatch than scallop dredges (Kaiser *et al.*, 1996). Analysis showed that both gears modified the benthic community in a similar manner, causing a reduction in most epifaunal species. A substantial amount of research in recent years has focused on increasing species selectivity in beam trawls to reduce unwanted bycatch. Revill & Jennings (2005) found that by incorporating benthic release panels into beam trawl nets, invertebrate bycatches were reduced by 75-80% and that greater than 90% of the animals released survived. Bergman & Van Santbrink (2000) found that, when considering fishing with beam trawls, the greatest amount of mortality is left on the seabed rather than occurring as bycatch.

Animals damaged by beam trawling rapidly attract scavengers (Sewell & Hiscock, 2005). Highly mobile scavengers, such as fish and crabs, quickly arrive at beam trawl tracks within minutes to hours, dispersing once feeding has taken place (Kaiser & Spencer, 1996). Whelks (*Buccinum undatum*) have been shown to survive beam trawling and can exploit a wide variety of prey, feeding on damaged and moribund animals in trawled areas (Evans *et al.*, 1996). Fish such as gurnard, whiting and dogfish, and the sea urchin *Strongylocentrotus pallidus*, are also known to aggregate over beam trawl tracks to feed (Kaiser & Spencer, 1994).

3.3.4 Recovery of habitats and communities

Changes which had taken place to megafaunal benthic communities from two different habitats (one with stable sediments and a rich fauna; the other with mobile sediment and a relatively impoverished fauna), six months after beam trawling had taken place, were assessed by Kaiser *et al.* (1998). They found that immediately after fishing, the stable sediment community was significantly altered: the abundance of some species had decreased (e.g., the sea mouse *Aphrodita aculeata*), while others had apparently increased (e.g., the hermit crab *Pagurus bernhardus*), although there was considerable variation between samples. This suggested the effects of trawling were not uniform. For the mobile sediment, no effects of trawling were apparent. After six months, the effects of any trawling disturbance were no longer evident.

In their review of UK fishing practices within UK European Marine Sites, Sewell & Hiscock (2005) point out that areas which have been intensively trawled for several years still support profitable fisheries which would not be possible without ample benthic food. It has therefore been suggested that it is not unlikely that the benthic community in these areas has shifted towards a dominance of highly productive, opportunistic species such as polychaetes (Rijnsdorp *et al.*, 1998; Jennings & Kaiser, 1998).

4. Beam trawling (for shrimp) on subtidal mud and subtidal sand

4.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports, the majority of which relate to UK waters. However, some research comes from the south-east North Sea and studies undertaken elsewhere. During the review of evidence, it has not always been possible to identify the precise gear being used, or the sediment type the gear is being dragged over, to be able to characterise the impact that results.

4.2. Background

In the UK, lightweight beam trawls are used to catch brown shrimp *Crangon crangon* (and, to a much lesser extent, pink shrimp *Pandalus montagui*) on muddy and muddy sand seabeds, typically using paired gears and in shallow waters (less than 15 m deep). In some locations (such as Morecambe Bay), lightweight beam trawls may also be deployed behind tractors operating in areas around the low water mark, as opposed to from boats. This method of fishing is not dealt with specifically here, although several of the impacts are likely to be similar. Unlike flatfish beam trawls, lightweight gears do not use chain mats or tickler chains, and as a result have less of an impact on the seabed and its associated communities.

There are few studies into the physical impacts of shrimp beam trawls on benthic habitats and communities, possibly because they are a relatively unusual gear type, but also probably because they are relatively lightweight and generally considered to have low levels of impact (ABPmer, 2015). One study (Paschen *et al.*, 2000) which looked at two types of shrimp beam trawls (fitted with a ground rope and rubber bobbins or chains), conducted in the Baltic Sea, found there was no significant sediment restructuring.

4.3. Impacts

4.3.1. Abrasion and penetration (to the physical habitat and damage to epifauna and infauna)

Beam trawling has been shown to flatten and homogenise the seabed through disturbance of the sediments (Tiano *et al.*, 2020). In muddy sediments, disturbance leads to larger changes in the biogeochemistry (due to the greater role of macrofauna-mediated processes here) compared to in sand (where hydrodynamics mediates the redox system) (Sciberras *et al.*, 2016). Nutrients, other chemical substances, and pollutants may also be released (Eigaard *et al.*, 2016). In The Wash shrimp fishery off England's east coast, the

abrasion/disturbance of the sediment has been identified as one of the main impacts of beam trawling. The primary impact results from the heavy trawl shoes and, to a lesser extent, by the net and the rollers (if used) (Eastern IFCA, 2018). The width of the shoes is about 20-40 cm, and they may leave a temporary track in softer seabed types e.g., muddy sand. However, traces of such tracks soon disappear in most cases due to the dynamic action of waves and tides (Addison *et al.*, 2017). Paschen *et al.* (2000) also found penetration by the trawl shoes was very shallow.

4.3.2. Changes in suspended solids; smothering and siltation

The passing of a beam trawl on the seabed will cause sediment to be re-suspended (Grieve *et al.*, 2014). This is particularly evident in deeper water where significant sediment plumes caused by demersal trawling are apparent and organisms are relatively unaffected by storm-related bottom stresses (Kaiser *et al.*, 2002). This re-suspended sediment can contribute to the total suspended sediment load, reducing light levels on the substrate (Jones, 1992). Subsequent deposition of the suspended sediment can also result in the smothering of feeding and respiratory organs of sessile organisms, potentially affecting biota in a wide area as the sediment is transported in the water column (Jones, 1992; Kaiser *et al.*, 2002). A study undertaken by ABPmer (2015) in The Wash identified the re-suspension of sediment as being an impact of beam trawling for shrimp, adding to the naturally high turbidity of the site. Modelling indicated that even within deeper parts of The Wash where there is less natural disturbance, surface sediments are mobile for 30-40% of the time (ABPmer, 2015).

4.3.3. Removal of species

Shrimp trawling is a non-selective fishing technique, with large numbers of juvenile fish species caught as bycatch, although various gear modifications have been developed to minimise this (Lüdemann & Koschinski, 2014). The capture of non-target species is identified as being one of the main impacts of beam trawling for shrimp in The Wash (Eastern IFCA, 2018). Elsewhere, large bodied infaunal organisms, such as the burrowing mud shrimp *Callinassa subterranea*, have been found to retreat deeper into their burrows to avoid beam trawls (Tiano *et al.*, 2020).

In a review of the impacts of chronic bottom trawling on different benthic habitats, Queirós *et al.* (2006) found that chronic trawling had a negative impact on the biomass and production of benthic communities in muddy habitats, while no impact was identified on benthic communities in sandy habitats. Off the coast of the Netherlands (the Frisian Front), Tiano *et al.* (2020) found that beam trawling simplifies the benthic foodweb by reducing the number of epifaunal organisms and shallow borrowers and found that the macrofaunal density of the surface-dwelling organisms was lowered by up to 74%. Species residing on or near the sediment surface, such as newly settled juvenile infauna, were found to be particularly vulnerable to trawling disturbances (Tiano *et al.*, 2020).

ABPmer (2015) determined that the biotopes present within The Wash are characteristic of naturally disturbed environments, dominated by infaunal species with high population

growth rates which can recover rapidly from disturbance. They found that the removal of brown shrimp is likely to be the most important ecological impact of the brown shrimp fishery due to the functional importance of *Crangon* as a predator and foraging species. Their prey includes bivalve spat and polychaete worms. Thus, the actual presence of brown shrimp can have a greater effect on the abundance of macrofauna than the physical disturbance resulting in changes to sediment composition (ABPmer, 2015).

Regarding discards, Lancaster & Frid (2002) found that, in the Solway Firth shrimp fishery, 99% of discarded undersized shrimps (once sorted on board) were returned to the sea alive and that after 24 hours it was estimated that 92% would have survived.

4.3.4. Recovery of habitats and communities

Long-term changes and potential recovery of the macrofauna are not only dependent on the frequency and scale of trawling but also on the nature of the sediments and the existing resilience to natural disturbances (Collie *et al.*, 2000; Kaiser *et al.*, 2002).

Slow-growing or fragile macrofaunal species living in the upper layer of sandy sediments, such as certain bivalves (e.g., *Spisula* sp.), echinoderms (e.g., *Echinocardium cordatum*) and tube-forming polychaetes (*Terebellidae* spp.) are known to be highly vulnerable to frequent trawling activities and have declined in abundance in such areas over time (Bergman & Hup, 1992; Jennings *et al.*, 2001).

5. Heavy otter trawling on subtidal coarse sediment and subtidal mixed sediment

5.1. Evidence

The evidence given below has come from peer-reviewed scientific papers, published reports and on-line documents. Most current available evidence for impacts of trawling on subtidal sediment focuses on subtidal sand, with few studies considering impacts on subtidal mixed or coarse sediments.

5.2. Background

Heavy otter trawls are those that use any of the following: sheet netting of greater than 4mm twine thickness; rockhoppers or discs of 200mm or above diameter; a chain for the foot/ground line (instead of wire); multiple tickler chains. Light otter trawls are those that use anything less than the definition of a heavy otter trawl (MMO, 2014).

5.3. Impacts

5.3.1. Abrasion and penetration (to the physical habitat)

The level of disturbance to the seabed from trawling gear will depend upon gear type, sediment type, trawling intensity, and natural processes (van Denderen *et al.*, 2015). Otter doors (also known as 'boards'), whilst designed to keep the mouth of the trawl open, will make contact with the seabed and in some fisheries this seafloor contact is crucial in creating a sediment plume which herds fish into the net (Grieve *et al.*, 2014). The doors penetrate deepest into sediments when compared with other elements of a single rig trawl, resulting in distinctive furrows and berms (Nilsson and Rosenberg, 2003; Polet & Depestele, 2010; Grieve *et al.*, 2014). The depth of ingress into the substrate is dependent on numerous factors, including the weight of the boards, the towing angle (determined by the length of warps and the speed of the towing vessel) and the composition of the sediment (Jones, 1992). Grieve *et al.* (2014) estimated an average penetration depth of 8.4 cm (no range given); other studies report furrows up to 30 cm deep (Jones 1992; Nilsson & Rosenberg, 2003; Queirós *et al.*, 2006); and Eigaard *et al.* (2016) recorded otter doors penetrating up to 10 cm in coarse and mixed substrata. Of all the bottom contacting components of an otter trawl, the doors affect the smallest area of the seabed, producing furrows up to a maximum width of only a few metres (Polet & Depestele, 2010), though they can produce the severest impact (Eigaard *et al.*, 2016).

Besides the otter boards, other elements of the gear which contact the bottom also affect the physical topography of the seafloor (Kaiser *et al.*, 2002; Eigaard *et al.*, 2016). Although not designed to penetrate the sediment, the ground rope, ground gear and associated

sweeps and bridles can skim along the surface of the seafloor flattening out topographic features (Jones, 1992; Nilsson & Rosenberg, 2003; Polet & Depestele, 2010). Where the ground rope travels above the sediment, the turbulence it causes can still affect the sediment, whilst high-relief topographic features will be damaged (Jones, 1992; Kaiser *et al.*, 2002; Polet & Depestele, 2010). The sweeps may also penetrate the seabed (likely to be just the top centimetres) and though they may have the least impact, they represent a large proportion of the total trawl gear path (Eigaard *et al.*, 2016). Additionally, many demersal trawls use bobbins or tickler chains attached to the ground rope to encourage fish into the net. These are heavier and likely to disturb the sediment, penetrating to a depth of between 2-5 cm (Eigaard *et al.*, 2016).

The remainder of demersal trawl gear is not designed to make bottom contact. However, elements such as the netting may make contact with the seabed if the cod-end is weighed down with rocks or trapped fish (Polet & Depestele, 2010). This is only likely to penetrate the sediment 0.1-1.8 cm, but may cause significant surface abrasion (Grieve *et al.*, 2014). Overall, when the ground gear and sweeps are included, the width of seabed affected by a single bottom otter trawl can vary between 25 m and 250 m (Eigaard *et al.*, 2016).

5.3.2. Changes in suspended solids; smothering and siltation

The action of bottom-contacting elements of heavy otter trawls causes sediment to be re-suspended (Kaiser *et al.*, 2002; De Madron *et al.*, 2005; Lucchetti and Sala, 2012; Grieve *et al.*, 2014). This is particularly evident in deeper water, as the organisms are relatively unaffected by storm-related bottom stresses and significant sediment plumes caused by demersal trawling are apparent (Kaiser *et al.*, 2002). As finer particles will settle more slowly than the larger particles and may be transported further away from the trawl track by the prevailing bottom currents, trawling will influence the sorting of the sediments in trawled areas (Brown *et al.*, 2005). The re-suspended sediment will contribute to the total suspended sediment load, reducing light levels on the substrate (Jones, 1992) and even enhancing phytoplankton primary production due to higher nutrient loads (Sciberras *et al.*, 2016; O'Neill & Summerbell, 2011). Subsequent deposition of the suspended sediment may also result in the smothering of feeding and respiratory organs of sessile organisms, as the sediment is transported in, and gradually deposited from, the water column (Jones 1992; Kaiser *et al.*, 2002).

Sediment re-suspension may also release nutrients and pollutants (such as heavy metals) held in the sediment, expose substrata with a reduced oxygen content, release contaminants and increase biological oxygen demand (Kaiser *et al.*, 2002; Queirós *et al.*, 2006; Bruns *et al.*, 2020). Both sediment re-suspension and mixing, as well as removal of infauna from the sediment by trawling, will affect the biogeochemical processes within soft sediment habitats.

5.3.3. Removal of species

Trawling may reduce benthic community biomass and biodiversity, and shift the assemblage composition towards short-lived, smaller species due to taxonomic

differences in direct mortality and recovery rates (Jennings *et al.*, 2005; Tillin *et al.*, 2006). The comprehensive reviews by Collie *et al.* (2000) and Kaiser *et al.* (2006) show how mortality imposed by the passage of a trawl is habitat specific and differs between benthic species groups and types of trawl gear. Additionally, the model developed by Hiddink *et al.* (2006) predicted the first pass of a trawl resulted in the most impact; subsequent passes had little additional effect on production and biomass.

In a global study, Hiddink *et al.* (2017) found that otter trawls removed an average of 6% of faunal biomass per pass (far less, as it happened, than hydraulic dredges which were found to remove 41% of biota); and in assessing the impacts of different gear types on benthic communities whilst targeting different species within the North Sea, Rijnsdorp *et al.* (2020) found that otter trawling for crustaceans had the highest impact, followed by otter trawling for demersal fish and beam trawling for flatfish and flyshooting.

Considerable decreases in the abundance of large and long-lived sessile fauna (e.g., erect sponges, fan corals, hydroids, erect bryozoans etc.) have been demonstrated to be a consequence of otter trawling over coarser sediments (Løkkeborg, 2004; Rijnsdorp *et al.*, 2018).

Trawling may dislodge benthic taxa anchored in soft sediments or displace taxa attached to hard substrate into unfavourable positions, while on harder substrates trawling may dislodge stones from the sediment by the action of tickler chains, rakes, or footrope, and these may subsequently be turned over, or end up in the net and be displaced or even removed (Auster *et al.*, 1996; Thrush & Dayton, 2002; Buhl-Mortensen *et al.*, 2013). Gear components may crush or break biogenic structures or material, such as dead shells, which may result in a reduction in the substrate for surface-dwelling species (Collie *et al.*, 2000; Kaiser *et al.*, 2006).

5.3.4. Recovery of habitats and communities

The recovery time of biogenic and geomorphological features impacted by fishing gear is dependent on the sediment type and the degree of natural disturbance which effects sediment stability, including physical (tidal current and wave actions), chemical and biological activities (Jones, 1992; Dernie *et al.*, 2003; Grieve *et al.*, 2014). Dynamic habitats in nearshore coastal zones are likely to recover quickly, sometimes within just a few hours (Jones, 1992; Grieve *et al.*, 2014), as are offshore areas with high natural disturbance due to having low initial species biomass (Hiddink *et al.*, 2006) and/or the benthic fauna being adapted to natural disturbance (Queirós *et al.*, 2006). More stable habitats (likely to be present in deeper waters or in shallow, sheltered locations) are likely to be affected by the impacts of fishing gear for longer (Kaiser *et al.*, 2002; Dernie *et al.*, 2003). The effects of trawling vary markedly among benthic species because of their different vulnerability to a trawl pass and different recovery rates following impact, varying from months to many years (Kaiser *et al.*, 2018). Recovery rates depend on recruitment of new individuals, growth of surviving biota, and active immigration from adjacent habitat. Most existing estimates of recovery rates come from experimental studies, with

abundances recorded both before and after experimental trawling takes place (Collie *et al.*, 2000; Kaiser *et al.*, 2006).

General studies have found that long-living, sessile and suspension feeding organisms show the greatest declines in response to a given type and frequency of trawl disturbance, while opportunistic species, e.g., short-living polychaetes, are less affected (Kaiser *et al.*, 2002).

6. Heavy otter trawling on subtidal sand and subtidal mud

6.1. Evidence

This review is based on information sourced from peer-reviewed journals and research reports, the majority of which relate to UK waters. However, some evidence comes from global reviews.

6.2. Background

Many elements of demersal otter trawl gear may come into direct contact with the seabed during some or all the fishing period, including the ground gear, sweeps, bridles, warps, doors, and parts of the net bag (Jones, 1992; Polet & Depestele, 2010). This may result in multiple physical effects on the benthic habitats and communities including (as described by Jones, 1992): scraping and ploughing (now referred to as abrasion and penetration); sediment re-suspension (smothering and siltation rate changes); and physical destruction or removal of non-target species (Nilsson & Rosenberg, 2003; Hall *et al.*, 2008).

6.3. Impacts

6.3.1. Abrasion and penetration (to the physical habitat and damage to epifauna and infauna)

Otter doors (also known as 'boards') are designed to keep the mouth of the trawl open and in so doing will drag across the seabed. In some fisheries this seafloor contact is crucial in creating a sediment plume which herds fish into the net (Grieve *et al.*, 2014). As a result, of all the major components of the otter trawl, the otter boards cause the deepest sediment disturbance in single rig trawls, creating distinctive furrows and berms (Sanchez *et al.*, 2000; Nilsson & Rosenberg, 2003; Polet & Depestele, 2010; Grieve *et al.*, 2014). The depth of furrows created by otter boards is highly variable, from a few centimetres to 30 cm deep (Jones 1992; Nilsson & Rosenberg, 2003; Queirós *et al.*, 2006; Grieve *et al.*, 2014; Bruns *et al.*, 2020). The depth of ingress into the substrate is dependent on numerous factors, including the weight of the boards, the towing angle (determined by the length of the warps and the speed of the towing vessel) and the composition of the sediment, with deeper furrows being recorded in soft mud (Jones, 1992). However, of all the major components of an otter trawl, the doors affect the smallest area of the seabed, producing furrows up to a maximum width of only a few meters (Polet & Depestele, 2010).

The persistence of track marks by trawl doors may last from several days or weeks (e.g., in nearshore coastal zones) to more than 18 months (e.g., in sheltered areas such as sea lochs or the deep sea), depending on the depth, sediment type, currents, wave action and

biological activity (Lindeboom & de Groot, 1998; Grieve *et al.*, 2014; Bruns *et al.*, 2020). Where tyres or rubber discs are incorporated into the leading edge of the trawl (known as a 'rock hopper'), these can create a series of furrows too.

Between the otter boards, the ground rope also effects the physical topography of the seafloor (Kaiser *et al.*, 2002). Although not designed to penetrate the sediment, the ground rope, bridles, and sweeps can skim along the surface of the seafloor flattening out high relief topographic features and causing turbulence (Jones, 1992; Nilsson & Rosenberg, 2003; Polet & Depestele, 2010). Heavy otter trawls typically use additional ground gear such as bobbins or tickler chains attached to the ground rope to encourage fish into the net. These are heavier and likely to disturb the sediment, penetrating an average of 1.8 cm into the benthos for gear greater than 24 m in width (Grieve *et al.*, 2014).

The warp and net of heavy otter trawl gear are not designed to make bottom contact. However, elements of the netting may make contact with the seabed if the cod-end is weighed down with rocks or trapped fish (Polet & Depestele, 2010). This is only likely to penetrate the sediment 0.1-1.8 cm, but may cause significant surface abrasion (Grieve *et al.*, 2014).

The main impacts from abrasion and penetration pressures associated with heavy otter trawls are to the biological communities. These pressures have been associated with decreased biomass, decreased species richness and diversity, and changes in community structure in fished areas when compared with similar unfished grounds (Ball *et al.*, 2000; Kaiser *et al.*, 2002). On a sandy bottom (on the Grand Banks off Newfoundland), Prena *et al.* (1999) found the biomass of the benthic species was 24% higher at sites with no trawling. However, a world-wide study by Kaiser *et al.* (2006) found no detectable initial impact from otter trawling on communities in sand habitats, whether assessed by total number of species or by individuals. Examining deposit feeders and suspension feeders separately also showed no detectable impact (Kaiser *et al.*, 2006).

6.3.2. Changes in suspended solids; smothering and siltation

The action of the bottom-contacting elements of heavy otter trawls on the seabed causes sediment to be re-suspended (Kaiser *et al.*, 2002; Grieve *et al.*, 2014). This is particularly evident in deeper water, as the organisms are relatively unaffected by storm-related bottom stresses and the significant sediment plumes caused by demersal trawling are apparent (Kaiser *et al.*, 2002). This re-suspended sediment can contribute to the total suspended sediment load, reducing light levels on the substrate (Jones, 1992). Subsequent deposition of the suspended sediment can also result in the smothering of feeding and respiratory organs of sessile organisms, potentially affecting biota over a wide area (the extent of which will depend upon the strength of the current at time of disturbance) as the sediment can be transported in the water column prior to settling (Jones 1992; Kaiser *et al.*, 2002). However, there would appear to be no direct evidence that changes in suspended solids, smothering and siltation pressures associated with fishing activities have detrimental impacts on the biological communities of sand or mud habitats.

Sediment re-suspension is likely to have a variety of effects including the release of nutrients held in the sediment, exposure of anoxic substrata, release of contaminants and increasing the biological oxygen demand (Kaiser *et al.*, 2002; Queirós *et al.*, 2006). Both sediment re-suspension and mixing, as well as removal of infauna from the sediment by trawling, will affect the biogeochemical processes within soft sediment habitats.

6.3.3. Removal of species

Subtidal muddy habitats are generally areas with low levels of natural disturbance and high levels of sediment deposition, which support a high density of infaunal communities (Ball *et al.*, 2000; Hall *et al.*, 2008). Macrofauna and near-surface infauna within these sediments are susceptible to disturbance from otter trawls (Ball *et al.*, 2000; Kaiser *et al.*, 2002; Queirós *et al.*, 2006). In addition, ecological impacts can include decreased biomass, decreased species richness and diversity, and changes in community structure (Kaiser *et al.*, 2002; Hiddink *et al.*, 2006). Queirós *et al.* (2006) found that chronic trawling had a negative impact on the biomass and production of benthic communities in muddy habitats, while no impact was identified on benthic communities from sandy habitats.

6.3.4. Recovery of habitats and communities

The recovery time for habitats impacted by fishing gear is dependent on the gear type (width/penetration); habitat type; and the degree of natural disturbance, e.g., physical (tidal current and wave actions), chemical processes, and biological activities (Jones, 1992; Dorn *et al.*, 2003; Foden *et al.*, 2010; Grieve *et al.*, 2014). Dynamic habitats in nearshore coastal zones are likely to recover quickly, sometimes within just a few hours (Jones, 1992; Grieve *et al.*, 2014), while more stable muddy habitats can be affected by the impacts of fishing gear for longer (Kaiser *et al.*, 2002; Dorn *et al.*, 2003). In the long-term, communities in areas of high natural disturbance are typically more resilient to the increased mortality rates generated by bottom trawling (Queirós *et al.*, 2006). Additionally, organisms with short life cycles, such as amphipods and polychaetes, will have relatively fast recovery rates, while slow-growing, longer-lived species, such as sponges and soft corals, will have longer recovery times (Roberts *et al.*, 2010; Kaiser *et al.*, 2006).

There have been few long-term studies of the impacts of otter trawls on muddy sediments, but those which have been conducted show highly varied recovery times. Some studies (e.g., Sanchez *et al.*, 2000; Kaiser *et al.*, 2006) have shown recovery to be relatively rapid (in a matter of days). Others have shown that gear tracks may well persist for at least 4 months (Smith *et al.*, 2007) or even for over a year (Tuck *et al.*, 1998); Sparks-McConkey *et al.* (2001) found infaunal assemblages in muddy sediments to have recovered after three and a half months.

Hall *et al.* (2008) considered the effects of gear types and fishing intensity on different habitat types. Subtidal stable muddy sands, sandy muds and muds were found to exhibit low sensitivity for low intensity demersal trawls (1-2 passes per month). However, at higher intensities, these habitats became increasingly sensitive, with high sensitivity to most demersal trawling activity at high intensity (daily passes). Foden *et al.*, (2010) found that

where otter trawling occurs too frequently on muddy sand, the habitat is unable to recover. In general, recovery rates given in the literature are variable from two days to in excess of a year.

7. Heavy Otter trawling on subtidal coarse sediment and subtidal mixed sediment

7.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports, the majority of which relate to UK waters. However, some research comes from Danish waters and studies undertaken elsewhere.

7.2. Background

Multi-rig trawls, introduced by the Danish fleet about 35 years ago and adopted by many others shortly afterwards, involve two or more otter trawl nets arranged side-by-side towed behind a single vessel. A clump weight between the nets is required to ensure the central part of the trawl remains in contact with the seafloor. Multi-rig trawls may be used to target Norway lobster *Nephrops norvegicus*, certain flatfish species such as Dover sole *Solea solea* and plaice *Pleuronectes platessa*, and roundfish species, such as cod *Gadus morhua*, and haddock *Melanogrammus aeglefinus*. They may also target rays in some regions of the UK. Feekings *et al.* (2014) found that whilst the introduction of multi-rig trawls to catch Norway lobster almost doubled the catch per unit effort over the period 1997-2012, there was no significant increase in the catch of roundfish species (such as cod, plaice, or haddock). This review deals primarily with the use of multi-rig trawls on muddy seabed habitats to catch Norway lobster *N. norvegicus*.

7.3. Impacts

7.3.1. Abrasion and penetration (to the physical habitat and damage to epifauna and infauna)

As multi-rig trawls are effectively multiple otter trawls, their impact on the seabed and the associated biota (both infauna and epifauna) is very similar to the impact of heavy otter trawls. However, a single multi-rig trawl will cover a wider area than a single otter trawl and will therefore produce a greater footprint on the seabed.

A review of experimental studies of the impact of towed fishing gears on benthic communities found that furrows and berms created by the trawl's doors are the most conspicuous physical impact caused by otter trawls on soft sediments, creating an irregular bottom topography (Løkkeborg, 2005). The penetration depth of the doors can be

as much as 35 cm in muddy sediments (Bruns *et al.*, 2020). With multi-rig trawls, the burrow and berm left by the clump weight is added to these impacts. As this weight may be up to 30% heavier than the (outer) otter doors, it will penetrate the sediment deeper than the rest of the gear to an average depth of 9.7 cm (range not stated) (Grieve *et al.*, 2014).

The depth of penetration into the substrate is dependent on numerous factors, including the weight of the gear, the towing angle, and the composition of the sediment, with deeper furrows being recorded in soft mud (Jones, 1992). However, of all the major components of an otter trawl, the doors and clump weight affect the smallest area of the seabed, producing furrows up to a maximum width of a few meters (Polet & Depestele, 2010). The remainder of the multi-rig trawl gear is not designed to make bottom contact, although elements such as the ground rope, ground gear, sweeps and netting may make contact with the seabed, particularly if the cod-ends are weighed down with rocks or trapped fish (Polet & Depestele, 2010; Grieve *et al.*, 2014). All these parts are only likely to penetrate the sediment 0.1-1.8 cm, but they are still likely to cause significant surface abrasion (Grieve *et al.*, 2014).

Subtidal mud habitats generally feature widespread, small scale, low relief topographic features such as ripples (Kaiser *et al.*, 2002; Grieve *et al.*, 2014). Habitat complexity is further enhanced through bioturbation creating mounds, burrows, and polychaete tubes (Nilsson & Rosenberg, 2003; Grieve *et al.*, 2014). Sediment penetration from otter boards creates furrows with much greater topographic relief than is normally present in these habitats, whilst abrasion flattens out small-scale topography, reducing the habitat complexity (Kaiser *et al.*, 2002; Nilsson & Rosenberg, 2003; Polet & Depestele, 2010; Grieve *et al.*, 2014). This results in a smooth seafloor interspersed infrequently with high relief features created by the furrows (Kaiser *et al.*, 2002).

Trawls used to catch Norway lobster *N. norvegicus* on muddy sediments may cause extensive damage to erect epifauna such as sea pens and burrowing anemones (Sewell & Hiscock, 2005). Ball *et al.* (2000) found that areas unfished for *Nephrops* were found to have a higher species diversity, numbers of individual organisms and biomass than fished areas: 49 species were recorded from unfished areas as opposed to 19 at fished sites and large specimens of several molluscs and echinoderms were found to be present at unfished but not at fished sites.

7.3.2. Changes in suspended solids; smothering and siltation

The action of the bottom-contacting elements of multi-rig trawls on the seabed causes sediment to be re-suspended (Kaiser *et al.*, 2002; Grieve *et al.*, 2014). This is particularly evident in deeper water, as benthic organisms are relatively unaffected by storm-related bottom stresses and the significant sediment plumes caused by demersal trawling (Kaiser *et al.*, 2002). This re-suspended sediment can contribute to the total suspended sediment load, reducing light levels on the substrate (Jones, 1992). Subsequent deposition of the suspended sediment can also result in the smothering of feeding and respiratory organs of sessile organisms (Jones 1992; Kaiser *et al.*, 2002). However, there is currently no direct evidence that changes in suspended solids, smothering and siltation pressures associated with fishing activities have a detrimental impact on biological communities of mud habitats.

Sediment re-suspension is likely to have a variety of impacts including releasing nutrients held in the sediment, exposure of anoxic substrata, release of contaminants and increasing biological oxygen demand (Kaiser *et al.*, 2002; Queirós *et al.*, 2006). Both sediment re-suspension and mixing, as well as removal of infauna from the sediment by trawling, will affect the biogeochemical processes within soft sediment habitats.

7.3.3. Removal of species

Catch selectivity of multi-rig trawls is greater than that for single otter trawls due largely to the design of the nets' openings. Thus, for catching Norway lobster and flatfish species, the narrow opening is held closer to the seabed (with a headline height usually less than 750 mm) than it is for multi-rig trawls designed to catch demersal roundfish (Graham & Ferro, 2004). However, when trawling for Norway lobster, due to the smaller mesh size used in comparison with demersal fish fisheries, bycatch may be high (Graham & Ferro, 2004). However, the shorter sweeps (bridles) on multi-rig gear help to reduce the capture of non-target species¹ (Seafish.org., n.d.).

7.3.4. Recovery of habitats and communities

Foden *et al.* (2010) found that the recovery time of habitats was determined by gear width, gear penetration, fishing frequency and sediment grain size. The recovery time of biogenic and geomorphological features impacted by fishing gear is dependent on the sediment type and the degree of natural disturbance which affects sediment stability, e.g., physical (tidal current and wave actions), chemical processes and biological activities (Jones, 1992; Dornie *et al.*, 2003; Foden *et al.*, 2010; Grieve *et al.*, 2014). Dynamic habitats in nearshore coastal zones are likely to recover quickly, sometimes within just a few hours (Jones, 1992; Grieve *et al.*, 2014), while more stable muddy habitats can be affected by the impacts of fishing gear for longer (Kaiser *et al.*, 2002; Dornie *et al.*, 2003). Dornie *et al.* (2003) reported muds having an 'intermediate' recovery rate and muddy sand habitats having the longest recovery rates. Additionally, organisms with short life cycles, such as amphipods and polychaetes, have relatively fast recovery rates, while slow-growing longer-lived species such as sponges and soft corals have longer recovery times (Roberts *et al.* 2010; Kaiser *et al.*, 2006). In the long-term, communities in areas of high natural disturbance are typically more resilient to the increased mortality rates generated by bottom trawling (Queirós *et al.*, 2006).

Otter trawls are the predominant gear deployed on muddy habitats and, despite the persistence of trawl marks, benthic recovery after otter trawling has been shown to be relatively rapid by some studies (i.e., days) in this environment (Sanchez *et al.*, 2000;

¹ <https://www.seafish.org/responsible-sourcing/fishing-gear-database/gear/multi-rig-trawl-nephrops-triple-rig/>

Kaiser *et al.*, 2006; Bruns *et al.*, 2020). In general, recovery rates given in the literature vary from two days to over a year.

Hall *et al.* (2008) considered the effects of gear types and fishing intensity on different habitat types. Subtidal muds were found to exhibit low sensitivity for low intensity demersal trawls (1-2 passes per month). However, at high intensities, these habitats became increasingly sensitive, with high sensitivity to most demersal trawling activity at high intensity (daily passes).

8. Light otter trawling on subtidal sand and subtidal mud

8.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports, the majority of which relate to UK waters. The literature search has found limited evidence of the impacts of light otter trawling on subtidal sand or mud. This review, therefore, reflects the paucity of available evidence and metrics given do not necessarily reflect the impacts across the whole fishery.

8.2. Background

Light otter trawls normally have a ground rope of around 10 m with nets approximately 10 to 20 m long. Light otter trawls generally comprise: netting of less than 4 mm twine thickness with rockhoppers or discs less than 200 mm diameter. Wire instead of a chain will be used for the ground rope and they will not use tickler chains (MMO, 2014). The doors will usually be between 0.5 and 1.5 m in length.

8.3. Impacts

8.3.1. Abrasion and penetration (to the physical habitat and damage to epifauna and infauna)

Demersal otter trawl gear has direct contact with the seabed wherever the ground rope, chains and bobbins, sweeps, doors and any chaffing mats or parts of the net bag make contact with the seabed (Jones, 1992; Polet & Depestele, 2010). However, there are considerable variations in both size and weight of trawls, with the impacts of the gear varying accordingly (Løkkeborg, 2005). Tracks are created on the seabed, the size and depth of which are dependent on the weight of the doors (Jones, 1992), the degree of contact with the seabed and the speed the gear is towed (Thrush & Dayton, 2002). While otter doors have the greatest impact on sediment, they only make up a small proportion of the total width of the gear (Polet & Depestele, 2010). Jones (1992) listed the ways in which

gears affect the seabed as scraping and ploughing (now referred to as abrasion and penetration); sediment re-suspension (smothering and siltation rate changes); and removal or scattering of organisms (Nilsson & Rosenberg, 2003; Hall *et al.*, 2008).

In a global study, Kaiser *et al.* (2006) found that in sand habitats, otter trawling had no significant impact on the substrate although there was evidence of post-fishing disturbance. In both muddy sand and mud habitats there was found to be a significant initial effect (Kaiser & Spencer, 1996; Kenchington *et al.*, 2001; Roberts *et al.*, 2010).

A review of experimental studies on the impact of towed fishing gears on different seabed types found that furrows and berms created by the trawl doors are the most conspicuous physical impact caused by otter trawls on soft sediments, creating an irregular bottom topography (Løkkeborg, 2005). Otter boards are designed to help keep the trawl close to the seabed and in some fisheries this seafloor contact is crucial in creating a sediment plume which herds fish into the net (Grieve *et al.*, 2014). The otter boards cause the deepest sediment disturbance in single rig trawls, resulting in distinctive furrows and berms (Sanchez *et al.*, 2000; Nilsson & Rosenberg, 2003; Polet & Depestele, 2010; Grieve *et al.*, 2014). The depth of furrows created by otter boards is highly variable and dependent on numerous factors, including the weight of the boards, the towing angle, and the composition of the sediment, with deeper furrows being recorded in soft mud (Jones, 1992). Where the ground rope travels above the sediment, the turbulence it causes can still affect the sediment, whilst high-relief topographic features will be damaged (Jones, 1992; Kaiser *et al.*, 2002; Polet & Depestele, 2010).

The remainder of demersal trawl gear is not designed to make bottom contact. However, elements such as the netting may make contact with the seabed if the cod-end is weighed down with rocks or trapped fish (Polet & Depestele, 2010).

8.3.2. Changes in suspended solids; smothering and siltation

The action of the bottom-contacting elements of otter trawls on the seabed causes sediment to be re-suspended (Kaiser *et al.*, 2002; Grieve *et al.*, 2014). This re-suspended sediment can contribute to the total suspended sediment load, reducing light levels on the substrate (Jones, 1992). Subsequent deposition of the suspended sediment can also result in the smothering of feeding and respiratory organs of sessile organisms, potentially affecting biota over a wide area (the extent of which will depend on the current strength at time of disturbance) as the sediment can be transported in the water column prior to settling (Jones 1992; Kaiser *et al.*, 2002). However, there would appear to be no direct evidence that changes in suspended solids, smothering and siltation pressures associated with fishing activities have detrimental impacts on biological communities of sand or mud habitats.

Sediment re-suspension is likely to have a variety of effects including the release of nutrients held in the sediment, exposure of anoxic substrata, release of contaminants and increasing the biological oxygen demand (Kaiser *et al.*, 2002; Queirós *et al.*, 2006). Both sediment re-suspension and mixing, as well as removal of infauna from the sediment by trawling, will affect the biogeochemical processes within soft sediment habitats.

8.3.3. Removal of species

Removal of species has the potential to affect the spatial distribution of subtidal mud and sand communities, change the presence and abundance of typical species and change the composition of the associated benthic communities (North-Western IFCA, 2017). An organism's vulnerability to fishing activity depends on its physical characteristics (hard or soft bodied), its mobility (mobile or sessile) and its habitat (infaunal or epifaunal) (Mercaldo-Allen & Goldberg, 2011). Larger bodied, slow moving, fragile organisms are most vulnerable (Kaiser & Spencer, 1996). Additionally, the effects of trawling can have different impacts upon organisms with different methods of feeding; for instance, otter trawling has been shown to have the greatest impact on suspension feeders in mud and sand habitats (Kaiser *et al.*, 2006).

The use of bottom towed gear has been found to result in decreased biomass, decreased species richness and diversity and changes in community structure (Ball *et al.*, 2000; Hall *et al.*, 2008). This can result in trawling grounds showing low species and biomass numbers in comparison with similar unfished grounds (Ball *et al.*, 2000; Kaiser *et al.*, 2002). Collie *et al.* (2000) undertook an analysis of published research into fishing activity impacts on the seabed, based on 39 research projects undertaken previously. They found an average of 46% decrease in the total number of species within the study sites that were disturbed with bottom towed gear. Sanchez *et al.* (2000) found apparently contrary evidence that there was little impact to the infauna from repeat trawls over the same ground. They found the abundance for most major taxa (e.g., polychaetes, crustacea and molluscs) was similar between fished and unfished areas. For other taxa, there was a distinct difference in abundances, with higher abundances associated with the fished areas compared with the unfished areas, possibly due to natural variability or the attraction of fished areas to scavengers and predators.

8.3.4. Recovery of habitats and communities

The recovery time of biogenic features impacted by fishing gear is dependent on the sediment type and the degree of natural disturbance which affects sediment stability, including physical (tidal current and wave actions), chemical and biological activities (Jones, 1992; Dernie *et al.*, 2003; Grieve *et al.*, 2014). 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. These findings were supported by Kaiser *et al.*, (2006) who undertook a global assessment of the biological impacts of different fishing activities and found the recovery of muddy sands was predicted to take months to years, and that sand was predicted to take days to months.

Mussel dredging on subtidal mud and subtidal mixed sediment

9.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports. Considerable research into the impacts of mussel dredging on sediments has come from northern Denmark (Limfjorden). Far less has been published in such detail from UK waters.

8.4. Background

Bivalve reefs (in this instance, of blue mussels *Mytilus edulis*) serve an important role as ecosystem engineers through their influence on nutrient cycling, water filtration, habitat structure, biodiversity, and food web dynamics (McLeod *et al.*, 2014).

The dredging activities for wild and seeded mussels in subtidal areas may change marine ecosystems in relation to benthic organisms (Dolmer *et al.* 2001, Dolmer 2002, Neckles *et al.* 2005) and substrate (Dolmer 2002) and may induce cascade effects on higher trophic levels, including birds (Atkinson *et al.* 2010). Dolmer & Frandsen (2002) identified two ecosystem services provided by *M. edulis*: they are important for the transport of material and energy from the pelagic to benthic systems, and in the control of phytoplankton biomass.

9.3. Impacts

9.3.1. Abrasion and penetration (to the physical habitat and damage to epifauna and infauna)

The mussel dredging process forms 2–5cm deep furrows in the seabed, but Dolmer *et al.* (2001) found the sediment texture and the organic content of the sediment was not affected.

Mainwaring *et al.* (2014) pointed out that activities resulting in abrasion and disturbance can directly affect the mussels by crushing them. Holt *et al.* (1998) noted that dredging of mussels will remove the substratum along with the mussels and their associated fauna and flora. Denny (1987) and Mainwaring *et al.* (2014) highlighted that dredging is also likely to increase the vulnerability of the remaining mussels to storm damage through the weakening of byssal attachment and by creating patches in the bed.

9.3.2. Changes in suspended solids; smothering and siltation

Dredging for blue mussels has been shown to induce the re-suspension of sediment for a few hours after dredging, together with a minor reduction in oxygen and a minor increase in ammonia (Dyckjær *et al.*, 1995), although the total amount of resuspension of sediment through dredging was found to be of the same order of magnitude as one or two wind-induced sediment resuspension events in a year (wind velocity 15m/s). Holt *et al.* (1998) found that dredging causes temporary re-suspension of sediment which could potentially result in localised smothering which has been shown to result in a loss of mussel condition and increased predation by crabs. Extraction of live mussels will also reduce filtration capacity (Frandsen *et al.*, 2015).

9.3.3. Removal of species

The effects of dredging for blue mussels (*M. edulis*) on benthic macrofauna are generally thought to be short-term (McLaverty *et al.*, 2020). This same study found that dredging reduced community biomass and impacted species composition. However, these results were not uniform; it was shown that eutrophication and natural disturbance can partly mask the effects of dredging on benthic communities. Recent meta-analysis of trawling impact studies has shown biomass to be an effective indicator of trawl disturbance (Hiddink *et al.*, 2020). Dolmer & Frandsen (2002) found that taxa such as sponges, echinoderms, anthozoans, molluscs, crustaceans, and ascidians, all had a reduced density, or were not observed at all, four months after an area had been dredged, indicating the fishery has an obvious effect on the epifauna.

No information has been found on the removal of species *per se* other than that mentioned above on epifauna. For infauna, an investigation on the impact of mussel dredging in a brackish Danish sound found that, immediately after dredging, a significantly lower number of infaunal species were measured inside the mussel beds compared to control areas (Dolmer *et al.*, 2001).

Abrasion and sub-surface damage are likely to attract mobile scavengers and predators, including fish, crabs, and starfish, to feed on exposed dead and damaged *M. edulis* individuals (Ramsay *et al.*, 1998). Experimental dredging trials of blue mussels (*M. edulis*) by Dolmer *et al.* (2001) reported increased densities of scavenging brown shrimps *Crangon crangon* in the days immediately post-dredging. The shrimps were suspected of feeding on polychaetes exposed at the sediment surface.

9.3.4. Recovery of habitats and communities

Mussel dredging has been found to reduce the available substrate (shell debris, pebbles, and the live mussel matrix) for settlement or post-settlement survival (Dolmer & Frandsen, 2002; McLaverty *et al.*, 2020). Larval settlement and recruitment of many invertebrates are strongly influenced by substrate structure (Lapointe & Bourget, 1999), and the altered state of the substrate post-dredging has been shown to interfere with recruitment, growth, and survival of the associated fauna (Dolmer & Frandsen, 2002).

Mytilus spp. populations are considered to have a strong ability to recover from environmental disturbance (Mainwaring *et al.*, 2014). A good annual recruitment may allow a bed to recover rapidly, though this cannot always be expected due to the sporadic nature of *M. edulis* recruitment. Therefore, blue mussel beds were considered by Mainwaring *et al.* (2014) to have a 'Medium' resilience (recovery within 2-10 years).

Dolmer *et al.* (2001) found that a reduction in species numbers post-dredging lasted for at least 40 days. Hoffmann & Dolmer (2000) reported on the effect of a closure of a *M. edulis* fishery in northern Denmark. They found that the closure had no significant influence on the demersal fish and epibenthic fauna, suggesting the overall eutrophication (and accompanying oxygen depletion) may have been more of a dominant influence.

In a study undertaken in northern Denmark, Frandsen *et al.* (2015) found mussel dredging trawl tracks were still visible on the soft sediment seabed two months after dredging.

10. Oyster dredging on subtidal mixed sediment

10.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports. Whilst there is considerable information on the demise and current population status of *Ostrea edulis* in UK waters, there appears little information published on the impact of the oyster fishery itself. Research is currently known to be ongoing; this synthesis will be reviewed when that research is published.

10.2. Background

The native, flat, or European oyster (*O. edulis*) has suffered a substantial decline throughout the UK over the past 100 years or so. There have been several attempts to re-establish populations in recent years with mixed success. Where native oysters are found in large numbers (more than five per square meter (Haelters and Kerckhof 2009.)), they form beds made up of the oysters themselves, and their dead shells on subtidal sediments. Many marine species such as gastropod molluscs, crabs and sea urchins live amongst these beds, with some using them as a place of shelter whereas others attach themselves to the surface. The small-scale oyster fisheries use small, single, lightweight dredges. On the south coast of England, a mechanical dredge known as a ladder dredge is used, restricted to a maximum width of 1.5 m (Southern IFCA, 2016a). Native oysters are typically fished over the winter period (November to February).

O. edulis is associated with highly productive estuarine and shallow coastal water habitats on firm bottoms of mud, rocks, muddy sand, muddy gravel with shells and hard silt. In exploited areas, suitable habitat is often created in the form of 'cultch' (broken shells) and

other hard substrata. Natural oyster beds of the species *O.edulis* are found in estuarine areas from 0-6 m depth on sheltered but not muddy sediments, where clean and hard substrates are available for settlement. They have also historically occurred in deeper waters and offshore, down to 50m, for example in the North Sea and the eastern English Channel, but these beds are now mostly depleted (Dyer, 2019).

Dead oyster shell also makes up a substantial portion of the substratum and clumps of dead shells and oysters can support large numbers of ascidians (sea squirts/tunicates), several large polychaete species, including suspension-feeding polychaetes; and a turf of red seaweeds such as *Plocamium cartilagineum*, *Nitophyllum punctatum* and *Spyridia filamentosa* may also be present where light levels permit.

Within the UK, the native oyster *O. edulis* has been the subject of a UK Species Action Plan since 1999 (UKBAP, 1999). Only a few exploitable populations of native oysters exist around the UK: the Essex Estuaries; the Solent; the River Fal; Strangford Lough (Northern Ireland); and Loch Ryan (south-west Scotland) (Allison *et al.*, 2020), also north Kent. There are various projects currently underway to re-introduce them elsewhere (e.g., see [Native Oyster Network.org](http://NativeOysterNetwork.org)). In some parts of the UK (such as the Fal Estuary in Cornwall), oysters may only be fished from a sailboat using a hand-dredge, a supposedly inefficient catch method but one which may have been responsible for the fishery's survival (Long *et al.*, 2017).

The native oyster *O. edulis* is susceptible to a range of threats including pollution, invasive species, disease, and overfishing (Dyer, 2019). Note that other species of oyster are also present in UK waters: the Portuguese oyster *Crassostrea angulata* and the non-native Pacific oyster *Crassostrea gigas*. The Portuguese oyster is not commercially fished or farmed; the Pacific oyster is largely farmed; neither are included within this review.

10.3. Impacts

10.3.1. Abrasion and penetration (to the physical habitat and damage to epifauna and infauna)

Oyster dredging is known to cause abrasion and subsurface disturbance to the seabed. Dredging is known to remove hard substrata (shells, stones, and live oysters) required for oyster recruitment and juvenile oyster survival. More generally, the vulnerability of an organism is ultimately related to whether it is epifaunal or infaunal, mobile or sessile, and soft-bodied or hard-shelled (Mercaldo-Allen & Goldberg, 2011). Epifaunal organisms are subject to crushing or at risk of being buried; infauna may be excavated or exposed. Several studies have found soft-bodied, deposit feeding crustaceans, polychaetes and ophiuroids to be most affected by dredging activities (Constantino *et al.*, 2009).

10.3.2. Changes in suspended solids; smothering and siltation

No specific information has been found on the impact of oyster dredges on the re-suspension or settlement of sediment in the water column.

10.3.3. Removal of species

No specific information has been found on the impact of oyster dredges on non-target species *per se* other than that mentioned above on epifauna and infauna. The action of an oyster dredge will be to collect 'cultch' material (that is, dead and broken shells) at the same time as live oysters are being collected. This cultch material is most important in providing a suitable habitat for the settlement of oyster spat, so is typically thrown back overboard during the on-board sorting process (Long *et al.*, 2017).

10.3.4. Recovery of habitats and communities

Repeated dredging may cause permanent damage to the seabed, adversely affecting the ability of the habitats to recover and may ultimately lead to the loss of the oyster population (Southern IFCA, 2016a). The timescale of recovery for benthic communities and potential prey species largely depends on sediment type, associated fauna, and the rate of natural disturbance (Roberts *et al.*, 2010). In locations where natural disturbance levels are high, the associated fauna is characterised by species adapted to withstand and recover from disturbance (Collie *et al.*, 2000; 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). With heavy demersal gear, Kaiser *et al.* (2006) estimated subtidal muddy sand habitats would take several years to recover, but that subtidal sandy habitats would take days to months to recover.

Recovery of communities will depend on the constituent species. Long-lived bivalves will undoubtedly take longer to recover from disturbance than other species (Roberts *et al.*, 2010). Slow-growing, large biomass biota, such as sponges and soft corals, are estimated to take up to eight years to fully recolonise areas, whilst biota with short lifespans, such as polychaetes, are estimated to take less than a year (Kaiser *et al.*, 2006).

11. Clam dredging on subtidal sand and subtidal mixed sediment

11.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports, with most information coming from studies undertaken in UK waters.

11.2. Background

The term 'clam' here refers to several different species of bivalve mollusc, all of which have their own habitat preferences. Only those fished for in the subtidal are considered in this synthesis.

Razor clams (*Ensis* sp.), also known as razor fish, razor shell or spoots (in Scotland), inhabit sandy and shelly substrates from the lower limits of the tidal range to about 40 m depth. They are fast, deep burrowers and are heavily preyed upon by the edible (brown) crab *Cancer pagurus* (Tuck *et al.*, 2000). *Ensis siliqua* is the largest of the razor clams, growing up to 25 cm in length and with a mean age of 25 years.

Razor clams are harvested using adapted hydraulic cockle dredges, which appear to produce less physical impact to the seabed than suction dredges (Tuck *et al.*, 2000). The water-jet dredge operates by fluidising the sand immediately ahead of it and then extracting the *Ensis* from the substrate by means of a hollow blade that protrudes almost 30 cm into the sediment. The forward movement of the dredge blade through the fluidised sand lifts the *Ensis* upwards and backwards into a collecting cage, assisted by a backwards water jet (Tuck *et al.*, 2000).

Clam dredging takes place within the Solent and Southampton Water all year round, both in the intertidal and subtidal. The fishery predominantly targets the non-native Manila clam (*Ruditapes philippinarum*), although the activity also targets American hard-shell clam (*Mercenaria mercenaria*). Occasional catches of the indigenous grooved carpet shell clam (*Ruditapes decussatus*) also occur. A 'box dredge' is used here to catch clams - a metal dredge with a row of metal teeth which are towed through the sediment using a boat (Southern IFCA, 2016b). The teeth penetrate the sediment disturbing the buried clams which are subsequently caught and retained in the dredge.

11.3. Impacts

11.3.1. Abrasion and penetration (to the physical habitat and damage to epifauna and infauna)

Following the pass of a hydraulic dredge when fishing for razor clams (*Ensis* sp.), Tuck *et al.* (2000) found there were immediate impacts to the seabed, with visible, 2 m wide trenches being apparent afterwards. It was also found that turbidity increased locally: the finest fraction of silt was soon dispersed by tidal currents, with heavier fractions settling on the surrounding area.

Within Poole Harbour (Dorset), a novel 'pump-scoop' dredge has been developed by local fishermen following the introduction of the Manila clam *R. philippinarum* in the 1980s (Clarke *et al.*, 2018). Although this gear is used in the intertidal zone [and thus not covered by the remit of this review], it was noted that there was a noticeable loss of fine sediments following dredging operations. Clarke *et al.* (2018) also found that the organic content of fine sediments in heavily dredged areas was reduced but could not say if this might have been due to the more dynamic environment which existed at such sites.

Tuck *et al.* (2000) noted an immediate reduction in the number of species, the number of individuals and the biomass within fished tracks (probably due to removal and redistribution of all infauna), but measures of diversity showed no effects. Hall *et al.* (1990) noted similar results. Sessile epifauna are rarely encountered in sandy environments due to the mobile nature of the latter. Thus, dredge impacts on epifauna are regarded as being minimal over sandy habitats. On mixed sediment however, Mercaldo-Allen & Goldberg (2011) found 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. A few 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.

Tuck *et al.* (2000) found that most of the infaunal community was not greatly affected by the (hydraulic) dredge, being already morphologically and behaviourally adapted to a dynamic environment. Within a day of a pass from a dredge for razor clams, it was found that the proportion of polychaetes within the substrate had decreased whereas the proportion of amphipods had increased. Hall *et al.* (1990) noted that immediately following the use of a hydraulic dredge for *Ensis* sp., there was a reduction in many macrofaunal species.

11.3.2. Changes in suspended solids; smothering and siltation

Tuck *et al.* (2000) found there were immediate impacts to the seabed following the passage of the when fishing for razor clams (*Ensis* sp.) using a [hydraulic] dredge, turbidity

increased locally, although the increase was not measured. The finest fraction of silt was soon dispersed by tidal currents, with heavier fractions settling on the surrounding area.

11.3.3. Removal of species

Clarke *et al.* (2018) found that, following intertidal fishing by the 'pump-scoop' dredge within Poole Harbour, reductions in abundance of the target species (Manila clam *R. philippinarum*) of up to 95% occur in some areas throughout the open season. Clarke *et al.* (2018) also found significant changes to macrobenthic community structure resulted, specifically, in an increase in abundance of certain polychaetes and a decline in abundance of the bivalve *Abra tenuis*. In considering hydraulic dredging for *Ensis* sp., Hall *et al.* (1990) noted that the immediate effects following dredging included a reduction in the abundance of species.

Tuck *et al.* (2000) found the commonest bycatch species to be the heart urchin *Echinocardium cordatum*, of which 28% of individuals retained by the dredge were fatally damaged. Small-bodied bivalve species were not damaged but three larger species (*Lutraria lutraria*, *Arctica islandica* and *Mya truncata*) were vulnerable to damage (10-20%). It is supposed that damage to these bivalves occurred as they came into contact with the blade while still in the sediment. Mobile epifaunal species (the crabs *Carcinus maenas*, *Corystes cassivelaunus* and *Cancer pagurus*) were retained by the dredge but escaped being damaged (Tuck *et al.*, 2000).

Post-dredging, Tuck *et al.* (2000) found several crab species (but most noticeably the shore crab *C. maenas*) attracted to the dredge track to scavenge on material disturbed by the dredge. *Liocarcinus depurator* and *Necora puber* were also recorded.

11.3.4. Recovery of habitats and communities

The timescale of recovery for benthic communities and potential prey species largely depends on sediment type, associated fauna, and the rate of natural disturbance (Roberts *et al.*, 2010). In locations where natural disturbance levels are high, the associated fauna is characterised by species adapted to withstand and recover from disturbance. 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). Recovery periods of several years were estimated for muddy sands, which Kaiser *et al.* (2006) found to be particularly vulnerable to impacts of fishing activities. The recovery periods for sandy habitats were estimated to take from days to months (Kaiser *et al.*, 2006).

Tuck *et al.* (2000) found that the trenches left by the hydraulic dredge had started to fill after five days and were no longer visible after 11 weeks. However, the sediment within the fished tracks remained fluidised beyond this period.

12. Pots/creels (crustacea/gastropods) on high energy infralittoral rock, moderate energy Infralittoral rock, low energy infralittoral rock, high energy circalittoral rock and moderate energy circalittoral rock

12.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports. Considerable research into the impacts of potting on rock substrates has come from the UK and European UK waters; additional evidence has come from the US and Caribbean. While some metrics are presented to enumerate impacts, they do not necessarily reflect the impacts across the whole fishery. Note that chalk, and peat and clay exposures are not included in this literature synthesis and impacts will be different on those softer rocks.

12.2. Background

Rocky benthic habitats (reefs) provide structural complexity for many sensitive and diverse epifauna, and such habitats may be vulnerable to sporadic or prolonged impacts from fishing activities and gear types (Gray, 1997; Gray *et al.*, 2006; Kaiser *et al.*, 2006; Sangil *et al.* 2013; Sheehan *et al.*, 2017; Kaiser, 2014; Salomidi *et al.*, 2012). Consequentially, the short- and long-term and secondary effects are wide ranging; such impacts may reduce species composition, biomass, and diversity, or result in removal of key species leading to changes in ecosystem functionality and resilience over different timescales (Gall *et al.*, 2020).

12.3. Impacts

12.3.1. Abrasion to the physical habitat and damage to epifauna

Abrasion impacts from pots may occur during deployment, positioning (via dragging), soak time, movement and hauling. While the pressure is unlikely to impact the rock substrate itself, it may impact taxa associated with the rock habitat (Rees, 2018; Gall, 2016; AWFA, 2020). Direct abrasive contact may occur from the pot itself, the end weight and anchors (Hartnoll, 1998; Eno *et al.*, 2001) Hall *et al.*, 2008; Rees, 2018; Johnson, 2002); indirect impacts may occur from scour or the rubbing effects to epifauna caused by the associated pot ropes.

Rope movement during the soak was observed for 51% of soaks, though in most of these instances (46%) this movement was described as minimal, with no scour or species impacts observed (Gall *et al.*, 2020).

The total possible contact areas per pot per haul (i.e., the total distance travelled x area of base of pot) were significantly larger than seabed contact areas (i.e., the distance where a pot contacted the seabed x area of base of pot) for different pot types, but were larger for inkwell pots than for parlour pots (inkwell pot, total possible contact area = 7.16 m², seabed contact area = 3.51 m²; parlour pot, total possible contact area = 5.24 m², seabed contact area = 2.57 m² (Gall *et al.*, 2020).

Studies carried out in Lyme Bay into the impacts of the 2013/2014 storms, on comparable habitat, found significant reductions in abundance, diversity and richness after the storms, and significant impacts on selected indicator taxa (Gall *et al.*, 2020); Rees, 2018). A study of Caribbean lobster traps found that post-storm damage included abrasion, fragmentation and removal of corals and sponges with an associated reduction on benthic species cover (Lewis *et al.*, 2009) The true impacts of potting may therefore be compounded by impacts from storms (Lewis *et al.*, 2009).

Significant pot movement can occur during spring or neap tides, indicating that even weak weather events may cause abrasion. The degree of gear movement varies depending on the strength of the weather event, wave height and tidal strength. The area potentially impacted by such pot movements ranged from 53 to 115 m² per pot (mean = 85.8 m²) though in reality these values are likely to be smaller (Stephenson *et al.*, 2015). At a local scale, potting impacts on epibenthic assemblages and species richness may not be detected against the background of natural variability (Stephenson *et al.*, 2015).

Abrasion commonly results in clouding of the water suggesting epifaunal tissue removal (Rees, 2018; Hall *et al.*, 2008) as well as sediment disturbance. Long lived, slow growing, sessile epifauna e.g., erect or branching fauna such as sea chervil (*Alcyonidium diaphanum*), dead man's fingers (*Alcyonium digitatum*), red boring sponge (*Cliona celata*), pink sea fan (*Eunicella verrucosa*), frilled anemone (*Metridium senile*) and the Ross coral (*Pentapora foliacea*) are particularly vulnerable to such impacts (Coleman *et al.*, 2013; Jennings and Kaiser, 1998; Sheehan *et al.*, 2013b; Langmead *et al.*, 2010; Appledorn *et al.*, 2000). Impacts of abrasion are not well studied, but evidence suggests that species such as sponges and soft corals may be left vulnerable to disease after damage has occurred (Bavestrello *et al.*, 1997; Hiscock, 2007; Shester and Micheli, 2011).

In abraded *E. verrucosa* colonies, regrowth can occur within days (Hiscock, 2007); However, if the coenenchyma (the common tissue that surrounds and links the polyps) is damaged or scraped off, prompt recovery could fail leaving them vulnerable to colonization by epibiota or disease. Subsequent mechanical stress and susceptibility to weakening from the burrowing activities of epibiota could induce mortality (Bavestrello *et al.*, 1997).

Stephenson *et al.* (2017) commented that encrusting species are unlikely to be damaged as their size and shape enable them to withstand impacts from physical disturbance and abrasion. While damage does occur at regularly fished sites, it is unlikely to be significant, unless potting intensity is sufficiently high (defined as ~30 pots for 500 m² in Rees, 2018; medium potting is defined as 15 - 25 pots and low potting as 5 - 10 pots in experiments)

where some decline in abundance is shown, and the footprint of the contact area is relatively small (Blyth et al., 2004).

Rees (2018) challenges the perception that commercial potting is benign based on more recent observations at “normal” potting levels (i.e., set/haul two – three times a week in stable weather [summer months] and once a week in unstable weather [winter months]); many recovering species, subsets and selected indicator species showed no impact from above-normal levels of potting on relative abundance, species richness and assemblage composition; however, Ross coral (*P. foliacea*) under Low, Medium and High potting treatments, and the sea squirt (*Phallusia mammillata*) under Medium and High treatments showed impacts of potting activity on recovery after three years.

Elevated levels of potting effort, regularity of activity and spatial concentration contribute to observable impacts on biogenic reef habitats over multiple years and exhibit in-year and between-years seasonal variability. In addition, Rees et al. (2021) present evidence demonstrating negative effects of (crab and lobster) pot fishing on both the temperate reef building, benthic epibiota and associated organisms, and the fishery itself, and that this effect is pot-density dependent. This study also demonstrates that a pot density threshold exists for this fishery (medium density treatment of 15 - 25 pots per 0.25 km²), however the authors acknowledge that few of the reef taxa assessed showed a response to increasing pot density.

12.3.2. Removal of species

Experiments have shown that 14 of 18 taxa identified, suffered damage from pots; individuals of six taxa were removed from the reef (Gall et al., 2020). Pot hauling damaged or removed between 25 and 30% of observed epibenthic species, with broadly consistent patterns between pot types (Gall et al., 2020). Individuals of *Echinus esculentus*, *Holothuria forskali* and *Asterias rubens* were observed to roll or be moved out of the way by the pressure wave from a pot (Gall et al., 2020). Analysis from this study identified that, following pot haul, significantly more individuals remain undamaged (0.54 ind. m⁻² (3%)). Selected taxa for which more individuals were damaged than were undamaged by inkwell pots were:

- *C. celata* (damaged = 54%, undamaged = 45%)
- *E. verrucosa* (damaged = 54%, undamaged = 45%), and,
- *P. foliacea* (damaged = 82%, undamaged = 13%)

The taxa removed from the reef included two upright branching taxa (*A. diaphanum* and *A. digitatum*) and two taxa with large forms projecting from the reef (*C. celata* and *P. foliacea*) and *Dendrodoa grossularia*, which attached to the reef at its base (Marlin, 2006). Consistency was exhibited across pot types in those species groups either impacted or removed from the reef by hauling (Gall et al., 2020).

Species which characterise areas fished with static gear more closely represent fully functioning benthic rocky reef areas; greater abundance may be driven by biogenic habitat forming species e.g., hydroids, bryozoans, and soft corals, indicating wider ecosystem importance (Jennings and Kaiser, 1998).

12.3.3. Recovery of habitats and communities

Stephenson et al. (2017) found that where potting activity caused damage to erect species, the (low) frequency with which a pot would be expected to impact the same area twice means that species would be able to recover sufficiently between fishing events (recovery time given as 6-36 months). It is noted that other species, e.g., axinellids may take up to 25 years to recover and structurally complex species e.g., *Lithothamnion corallioides*, much longer.

Rees (2018) posited that potting is more destructive than originally thought and only cessation of potting activity would truly permit full recovery of species; experimental potting at different intensities found significant impacts on abundance and species richness (especially in high intensity potting areas) for *P. foliacea* and *P. mammillata* (Rees, 2018). Potting density will also be an issue to consider, and impacts may be more substantial in areas where (faunal) abundance is greater (Gall *et al.*, 2020), however this remains a gap in the evidence base. Rees et al. (2021) demonstrates impacts at higher potting densities and offers evidence to provide a threshold for such.

13. Gill nets, trammel nets and entangling nets on high energy infralittoral rock, moderate energy infralittoral rock, low energy infralittoral rock, high energy circalittoral rock and moderate energy circalittoral rock

13.1. Evidence

The evidence used in this review has come from peer-reviewed scientific papers, published reports and on-line documents, and comes from the UK, Europe, US, Canada, and Australia.

In the UK, Tillin *et al.* (2020) note the following evidence gaps for subtidal rock habitats' taxa for which little pressure/sensitivity information was available: sponges, ascidians, gastropod molluscs, and crustaceans which may be impacted by the gear elements.

13.2. Background

Static nets such as gill, trammel and entangling nets are widely used fishing methods (FAO, 2001; Petteta *et al.*, 2020; Williams *et al.*, 2011). When set or retrieved from the water, different elements of the gears (anchors, weights, and ground lines) can land heavily, or be dragged across the seafloor, impacting substrates and epifauna. While the area directly affected is small, epifaunal communities may be greatly impacted (Grieve *et al.*, 2014; Natural England, 2014). In areas with stronger currents, the net itself can be pushed down onto the seafloor and may snag on rock or branching structures (Grieve *et al.*, 2014). Improved materials have reduced potential for net damage and techniques have enabled these gears to be used over wrecks, rocky reefs, and deep water to fish several target species (Suuronen *et al.*, 2012; He and Pol, 2010).

Trammel nets are considered analogous with gillnets and entangling nets in terms of their impacts and vary only in design and set up (i.e., mesh size, sheet size, number) however subtle modifications (e.g., twine thickness, hanging ratios) can significantly reduce unwanted catch (Ford *et al.*, 2020).

13.3. Impacts

13.3.1. Abrasion (to the physical habitat and damage to epifauna)

Benthic impacts from this activity mainly occur during retrieval, with anchors and ground-lines coming into direct contact with the seabed (Grieve *et al.*, 2014). Different parts of the

gear can snag on demersal structures or fragile, sessile species (Johnson, 2002). While abrasion may cause sediment veneer disturbance, and damage to epifaunal/epifloral communities, physical damage to the rock itself is unlikely (Tillin *et al.*, 2010). Associated effects may be removal and/or displacement of organisms and structures, disturbance of sediment veneers which cover rock may cause light or temporary smothering (Tillin *et al.*, 2010). There is recent evidence of potential abrasion and physical damage to chalk rock in Cromer MCZ which is being investigated (Tibbitt *et al.*, 2020). If gear is dragged along the bottom before hauling the impact footprint will increase, (Grieve *et al.*, 2014). Gear may shift either by the current, wind or storms, thus damaging organisms on the seabed beyond the usual footprint of the gear; this action may also overturn cobbles and small boulders to which organisms may be attached (Grieve *et al.*, 2014). Damaged, abandoned, or lost gillnets may also have direct, indirect, and long-lasting effects (ghost fishing) and the physical effects on organisms may be like those caused during gear retrieval (Erzini *et al.*, 1997; Large *et al.*, 2009). Impacts may be exacerbated or prolonged where different fishing gears are co-located and interact causing negative and cumulative interactions (Kaiser, 2014).

13.3.2. Removal of species

As gill nets select for size of the target species (e.g., saithe, pollack, ling and cod - Cornwall Wildlife Trust, 2018) there is significant bycatch of fish and many other taxa. Removal of such non-target species may impact directly through degradation of ecologically important habitat or habitat-forming species (e.g., kelps, sponges, and corals), disruption of community structure and reduced productivity; or, indirectly through reduced species abundance or impaired ecosystem function (Shester & Micheli, 2011). Shester & Micheli (2011) explored potential impacts at two sites characterised by temperate to subtropical kelp forests and rocky reefs between 5 m and 22 m depth. Set gillnets showed the highest mean discard rates (34% by biomass, over 30 observed trips) when compared to other gears (lobster traps; fish traps; and drift gill nets). Use of gillnets is responsible for removal of non-target species including seabirds (Sonntag *et al.*, 2012; JNCC, 2012; Pedersen *et al.*, 2009; Sewell *et al.*, 2007) elasmobranch species (Cornwall Wildlife Trust, 2018) and mammals (Nielsen *et al.*, 2013; Pedersen *et al.*, 2009).

Set gillnets may also tangle and remove, kelp plants, gorgonian corals, sponges, and other branched, biogenic structures (Shester & Micheli, 2011). In their experiments from the US, they showed net gear to be in contact with the seafloor 43% of the time or within 2 m of the seafloor 53% of the time. Any interactions between nets and branched or habitat-forming species resulted in organisms being partially damaged or completely removed. Of 60 observed interactions of gillnets with kelp (*Eisenia arborea*), 27 resulted in full removal; 15 in partial damage, with 18 showing no visible damage. Of 22 coral interactions, eight resulted in full removal, nine in partial damage, with five showing no visible damage.

Set gillnets may damage or remove gorgonians through repeated activity, with the relative cumulative damage becoming increasingly severe (Shester & Micheli, 2011) when compared to seabed contact area for kelp, gorgonians, and hard corals from a single pass of a bottom trawl, (NMFS, 2005; Krieger, 2001); however, the contact area is magnitudes

of order lower for gillnets. Partial damage to gorgonians can facilitate harmful algal growth on the tissue scars, which may have long lasting effects or result in mortality (Van der Knapp, 1993) and potential for wider and cumulative ecosystems impacts (Shester & Micheli, 2011).

In Australia, Bell *et al.* (2016) notes that over half the commercial gillnet catch (deployed over rocky reef) was discarded, with rates of about 20% for target and more than 80% for non-target species. Capture condition, including initial mortality, was assessed for a range of species with bycatch mortality more likely with increased soak duration.

O'Brien and Dennis (2008) found in a comparative experiment conducted in Canada, the mortality for gillnet-caught Atlantic cod was low (less than 5%) at 6-hour soak time but raised to about 30% with 12-hour soak time and continued to increase with longer soak times. Poor handling, increased air and/or water temperatures and capture depth also increased mortality (Bell *et al.*, 2016). These factors are considered a problem in many NE Atlantic gillnet fisheries. Longer term bycatch impacts are also likely to be dependent on the life histories of the species concerned (Ford *et al.*, 2020).

13.3.3. Recovery of habitats and communities

In New Zealand, Dawson and Slooten (2005) observed that rocky reef environments in temperate waters support a wide variety of species that are dependent on the substratum, kelp and associated taxa for shelter and survival. Damage to habitats from gillnet fisheries therefore carries potential for longer term ecosystem impacts and imbalance in structure and dynamics of populations. Epifaunal and epifloral communities' recovery following gill netting activity is not well understood, however, Tillin *et al.* (2010) note that generally subtidal rock communities recover rather slowly after disturbance, displacement, or full removal of species from rock substrates. Recovery is dependent on the nature, extent and frequency of the disturbance and is species specific with some epifaunal species able to reattach (recovery within days/weeks) or may take many years e.g., recovery of knotted wrack (*Ascophyllum nodosum*) can be very slow, greater than 12 years (Jenkins *et al.*, 2004, *in* Tillin *et al.*, 2010).

14. Pots/creels (crustacea/gastropods) on subtidal macrophyte-dominated sediment and maerl beds

14.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports. There is a considerable evidence gap relating to the impacts of potting gear on Maerl in general.

14.2. Background

Maerl beds are living and dead aggregations of slow growing (cited variously as ~500 μm to a few mm per year) (Tauran *et al.*, 2020; Salomidi *et al.*, 2012) coralline algae and may be long lived (greater than 100 years) and with a wide depth range from intertidal sites to 270 m (Riosmena-Rodríguez *et al.*, 2016). They are bioengineers and provide a complex three-dimensional habitat for associated species (Riosmena-Rodríguez *et al.*, 2016; Tauran *et al.*, 2020) that aids recruitment of commercially important species. While resilient to a range of environmental fluctuations, maerl may be adversely impacted by fishing activities (Riosmena-Rodríguez *et al.*, 2016; Tauran *et al.*, 2020). Both living and dead maerl may provide the structural integrity and complexity required for associated surface-dwelling organisms, with mixed maerl beds supporting greater number of taxa and abundance in some areas; live maerl, however, exhibits greater heterogeneity (Sheehan *et al.*, 2015; Salomidi *et al.*, 2012).

Walmsley *et al.* (2015) state that at time of writing there was no primary evidence for the impacts of potting activity on maerl beds. While there has been more recent interest and sensitivity assessments exist (Hall *et al.*, 2008), there is an acknowledged lack of evidence for interactions between potting gear and maerl beds (Tillin *et al.*, 2010).

Maerl sensitivity assessments and much of the research are based on trawling or dredging activity (Walmesley *et al.*, 2015; Tauran *et al.*, 2020; Howarth *et al.*, 2015) and consider maerl's fragility and extremely slow growth rate (i.e., recovery time) in assigning sensitivity.

14.3. Impacts

14.3.1. Abrasion (to the physical habitat and damage to epifauna)

The intensity of potting activity is important and while an attempt has been made to define low medium and high potting rates over rock substrates (Rees, 2018), no similar evidence is available for potting intensity on maerl. The set/haul frequency will also be important as the cumulative abrasion pressure from frequent set/haul is not well understood and may have a detrimental effect (Rees, 2018; Rees *et al.*, 2021; Walmesley *et al.*, 2015). It should be regarded that further research is required for activities other than trawling and dredging.

As well as impacting many faunal taxa which utilise the structural complexity of maerl, abrasion pressure may also affect those seaweed species which are thought to be maerl specialists (i.e., *Cruoria cruoriiformis*, *Cladophora rhodolithicola* and *Gelidiella calcicola*) found in Britain, Ireland, France, and Spain (Peña *et al.*, 2014).

14.3.2. Removal of species

Hinz *et al.* (2009) suggest that commercially targeted species may partially depend on sensitive habitats such as maerl, which provides a nursery ground for species as cod,

saithe, and queen scallops. The structural complexity of maerl habitats may increase species richness and abundance for many species at different life cycle stages, e.g., provide a settlement substrate and predation refuge for juveniles (Hinz *et al.*, 2009). Whilst there is little evidence available, potting activity in areas of such richness and abundance may physically remove higher proportions of target species than non-target species due to the selectivity of potting gear.

14.3.3. Recovery of Habitat and Communities

The impacts of potting gear on maerl beds are not well understood owing to a paucity of evidence. The sensitivity assessments being based on slow growth rates and fragile nature of maerl suggests that any damage incurred will be immediate and long lasting. Recovery of the maerl itself may take decades from bivalve dredging activity (Tauran *et al.*, 2020), though dead thalli have structural and ecosystem roles to play in habitat provision for surface-dwelling organisms (Sheehan *et al.*, 2015) and community recovery is likely to be much more rapid.

Based on experimental trawling data only, the predicted community recovery rates have been estimated at about eight years for community abundance and more than 100 years for species richness (Tauran *et al.*, 2020).

15. Gill nets, trammel nets and entangling nets on subtidal macrophyte-dominated sediment and maerl beds

15.1. Evidence

The literature search found very little evidence of the impacts of netting on maerl. This review, therefore, reflects the paucity of available evidence which is sourced from peer-reviewed scientific journals and research reports.

15.2. Background

Maerl beds are living and dead aggregations of slow growing (cited variously as ~500µm to a few mm per year) (Tauran *et al.*, 2020; Salomidi *et al.*, 2012) coralline algae and may be long lived (greater than 100 years) and with a wide depth range from intertidal sites to 270 m (Riosmena-Rodríguez *et al.*, 2016). They are bioengineers and provide a complex three-dimensional habitat for associated species (Riosmena-Rodríguez *et al.*, 2016; Tauran *et al.*, 2020) that aids recruitment of commercially important species. While resilient to a range of environmental fluctuations, maerl may be adversely impacted by fishing activities (Riosmena-Rodríguez *et al.*, 2016; Tauran *et al.*, 2020). Dead maerl may also provide the structural integrity and complexity required for associated surface-

dwelling organisms, with mixed maerl beds supporting greater number of taxa and abundance in some areas; live maerl, however, exhibits greater heterogeneity (Sheehan *et al.*, 2015, Salomidi *et al.*, 2012). In the UK, the dominant species of maerl are *Phymatolithon calcareum*, *Lithothamnion glaciale* and *Lithothamnion corallioides* (MarLIN, 2006) and as a fragile species it has a very low resilience assessment and little to no ability to recover given this fragility and extremely slow growth rate (MarLIN, 2006). There is no evidence of sensitivity differentiation between these maerl species.

Walmsley *et al.* (2015) state that at time of writing there was no primary evidence for the impacts of gill/trammel/entangling net activity on maerl beds. While there has been more recent interest and sensitivity assessments exist (Hall *et al.*, 2008), there is an acknowledged lack of evidence for interactions between this gear type and maerl (Tillin *et al.*, 2010).

These sensitivity assessments and much of the research are based on trawling or dredging activity (Walmesley *et al.*, 2015; Tauran *et al.*, 2020; Howarth *et al.*, 2015) and consider maerl's fragility and extremely slow growth rate (i.e., recovery time) in assigning sensitivity.

15.3. Impacts

15.3.1. Abrasion (to the physical habitat and damage to epifauna)

Ongoing research on trawled maerl beds in the Columbretes area of Mediterranean Spain, has revealed larger sizes of *L. corallioides* in the established (25-year-old) areas of an MPA, but smaller sizes in the 6-year-old region of the MPA (Barberá *et al.*, 2017). This study also found that percentage cover was higher within the MPA, however, this was coupled with a lower proportion of live maerl within the same area. Higher cover and smaller sizes of *L. corallioides* were also found in an area with higher trawling intensity (Barberá *et al.*, 2012b). Such mechanical impacts of fishing can affect maerl shape and size structure (Barberá *et al.*, 2017). While the impacts of netting on maerl beds is not well documented however, *L. corallioides* may be more susceptible to fragmentation from mechanical impacts, such as would be expected from continuous trammel net activity (Foster, 2001). This repeated activity and disturbance may facilitate recruitment (through dispersal of live fragments) subsequently exhibiting increasing percentage cover of smaller rhodoliths in higher live quantities, detectable in the short-term (Barberá *et al.*, 2017).

Impacts from trammel-netting even when spatially concentrated, are unlikely to cause the type and degree of physical impacts as described for mobile, bottom towed gear (Barberá *et al.*, 2017), however, net anchors are likely to penetrate the complex structure itself or the substrate and potentially undermine maerl's structural integrity though no direct evidence of this was found. Hall-Spencer *et al.* (2017) noted that mooring chains attached to fish farm nets at various sites in Scotland also caused crushing damage to under-lying *P. calcareum* beds and shading and smothering by nets and associated ropes.

As well as impacting many faunal taxa which utilise maerl structural complexity, abrasion pressure may also affect those seaweed species which are thought to be maerl specialists (i.e., *Cruoria cruoriiformis*, *Cladophora rhodolithicola*, *Gelidiella calcicola*) found in Britain, Ireland, France, and Spain (Peña *et al.*, 2014).

15.3.2. Penetration and/or disturbance of the substratum below the surface of the seabed, including abrasion

It is acknowledged that netting anchors may cause abrasive and penetrative pressures, and when set on maerl beds the anchors may snag on the complex structures. While no specific evidence was found for netting on maerl beds, there is evidence of impacts from a range of trawling gear and aquaculture (Scottish Government, 2013; MarLIN, 2006) with impacts being framed in this context. Such physical disturbance may impact the structural integrity of the bed and break up loose nodules leaving them vulnerable to dispersal (Scottish Government, 2013). It is noted that repeated boat anchoring could create cumulative impacts normally associated with towed fishing gear, though no intensities or frequencies are given. Other consequential impacts of anchor deployment may include loss of habitat extent, temporary increases in suspended sediments and disruption of associated community structure and function.

15.3.3. Removal of species

Hinz *et al.* (2009) suggest that commercially targeted species may partially depend on sensitive habitats such as maerl, which provides a nursery ground for species as cod, saithe and queen scallops. The structural complexity of maerl habitats may increase species richness and abundance for many species at different life cycle stages, e.g., provide a settlement substrate and predation refuge for juveniles (Hinz *et al.*, 2009).

Specifically, trammel netting for spiny lobster is thought to impact maerl beds through snagging and breakage either through hauling or cleaning, or when poor weather conditions delay hauling and prolong soak times (Barberá *et al.*, 2017). These authors also note that no published study has addressed the impacts of trammel nets on maerl, and observations of Borg *et al.* (1998) persist (Barberá *et al.*, 2017). The selectivity of netting gear may mean that maerl associated finfish are targeted successfully, although there appears to be no evidence on bycatch from static netting activity on maerl beds.

15.3.4. Recovery of Habitat and Communities

The impacts of netting gear on maerl beds are not well understood owing to a paucity of evidence. The sensitivity assessments being based on slow growth rates and fragile nature of maerl suggests that any damage incurred will be immediate and long lasting. Recovery of the maerl itself may take decades from bivalve dredging activity (Tauran *et al.*, 2020) though dead thalli have structural and ecosystem roles to play in habitat provision for surface-dwelling organisms (Sheehan *et al.*, 2015) and community recovery is likely to be much more rapid.

Based on experimental trawling data, the predicted community recovery rates have been estimated at about eight years for community abundance and more than 100 years for species richness (Tauran *et al.*, 2020).

16. Pots/creels (crustacea/gastropods) and cuttle pots on subtidal seagrass beds

16.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports primarily from the UK, Europe, and US. There is an evidence gap relating to the impacts of potting and static gears on seagrass in general.

16.2. Background

Fishing gears are commonly used where seagrass is found either in the form of pots or as bottom set gill or trammel nets. Whilst the potential for damage is lower per unit deployment compared to towed gear, there is a risk of cumulative damage if use is intensive. The severity of impacts may depend on whether disturbance occurs above (to leaves and stems) or below the ground (to rhizomes) (Stephan *et al.*, 2000). There is no robust evidence to suggest that mechanical disturbance caused by fishing (or other) activity is beneficial, or that uprooted marine plants are likely to re-root themselves (Stephan *et al.*, 2000).

Hall *et al.* (2008) found that seagrass habitats within the UK exhibit a high sensitivity to “heavy” levels of potting activity (daily set/haul, density greater than 5 pots per ha), a medium sensitivity to moderate and low levels of potting (less than 4 pots per ha) and a low sensitivity to single pot usage; however, no direct evidence was demonstrated, and the study recommended further research and monitoring where potting intensity is moderate or high. This currently represents an evidence gap and an unquantified level of sensitivity to potting intensity, although Rees (2018) attempted to quantify potting density on rock substrates, (Low potting: 5-10 pots per 500 x 500 m; Medium potting: 15-25 pots per 500 x 500 m; high potting: greater than 30 pots per 500m x 500m area).

While there is no, or very limited, primary evidence (as in Walmesley *et al.*, 2015) on the impact of potting on seagrass directly, more recent research on potting intensity (Rees, 2018) may enable application to gear type and habitat combination for this activity. The type of pots most likely to be used on seagrass beds in the UK are cuttlefish traps.

Seagrass meadows provide complex habitats (Duffy, 2006), which support many taxa. In a study of Mediterranean and Atlantic *Zostera spp.* meadows, Mosbahi *et al.* (2017) recorded a total of 232 taxa, with annelids and molluscs the dominant groups at study sites.

16.3. Impacts

16.3.1. Abrasion (to the physical habitat and damage to epifauna)

Walmesley *et al.* (2015) suggest that pots which are consistently set and hauled provide potential for abrasive movement and subsequent, considerable damage to subtidal seagrass beds and their associated invertebrate fauna. This may present as leaf-shearing, meristem damage, uprooting, smothering, light reduction, and attenuation on seagrass (Roberts *et al.*, 2010; Stephan *et al.*, 2000; Jenkin *et al.*, 2017). While the individual plant cannot regrow damaged leaves or stems, it may produce new growth from intact meristems and non-UK species such as *Posidonia* spp. and *Halophila* spp. are more susceptible to above-ground impacts (Stephan *et al.*, 2000) and potentially, seagrass wasting disease which, historically, has significantly affected the extent of *Zostera* spp. beds in the UK (Tyler-Walters, 2008; Hughes *et al.*, 2018). Seed or flower shearing may be caused by static gear parts which move around in currents or storms, impacting the plant's reproductive ability. Loss of flowers or seeds will affect the subsequent years growth of new plants and impact would likely be most detrimental for species that rely heavily on sexual reproduction (e.g., *Zostera marina*) (Stephan *et al.*, 2000).

The nature of uprooting of plants (random and occasional) through snagging and displacement and without sediment disturbance may be caused by potting activity and is regarded as impacting seagrasses above-ground (Stephan *et al.*, 2000). Potting gears are unlikely to have below-ground impacts as they do not penetrate the sediment to the extent described in d'Avack *et al.* (2014).

Long pot soak times or ghost fishing from lost pots, may prolong these impacts or displace more fragile seagrass species (NOAA, 1996) though no primary evidence is provided. Several factors may combine to affect the degree of damage from potting such as the intensity of potting activity (number of pots), soak time, and set/haul frequency (JNCC and NE, 2011) with the potential for storms to exacerbate these impacts through increased pot movement; seasonality and intensity during sensitive spawning periods may also have impacts of structure, function, and recruitment (Stephan *et al.*, 2000). Sufficiently large numbers of pots, or cumulative use in an area could result in elevated levels of losses to submerged aquatic vegetation such as seagrass (Stephan *et al.*, 2000; Roberts *et al.*, 2010). Use of lobster traps in New England is considered a contributor to such vegetation loss (Walmesley *et al.*, 2015).

16.3.2. Removal of species

The Marine Evidence-based Sensitivity Assessment (MarESA) for seagrass states that this pressure is generally exerted by targeted fisheries for species associated with seagrass meadows (i.e., mussels, clams, and scallops), but that these are fished by trawling, dredging, digging, and raking. It is acknowledged, however, that seagrass plants and the sediment they grow in may be directly removed or damaged by static gears targeting other species. Any incidental removal of seagrass or associated species may

alter the character and structure of the biotope with consequential change in overall species richness and ecosystem services provision (d'Avack *et al.*, 2014).

16.3.3. Recovery of Habitat and Communities

Seagrasses form meadows that consist of one or only a few species that perform a wide variety of functions typically associated with numerous species in other systems (Reusch & Hughes, 2006). Duffy (2006) asserts that genetic diversity enhances seagrass growth and resistance to disturbance. More genetically diverse assemblages of *Z. marina* exhibit faster recovery after the summer mortality than less diverse assemblages, with more shoots, biomass, and associated invertebrates (Reusch & Hughes, 2006). Dominant, supporting species such as seagrasses, have bottom-up influences on associated or dependent communities, therefore, such species can influence the structure, functioning and recovery of entire ecosystems (Reusch & Hughes, 2006). Concomitant abrasion and crushing pressure may also intensify impacts and increase the risk of cumulative impacts over time with the risks likely to occur from potting activity leading to an extended recovery period (d'Avack *et al.*, 2014). Whilst there is no direct evidence for recovery periods from potting activity, it has been suggested that species such as turtle grass (*Thalassia testudinum*) may take between 5 and seventeen years to recover from propeller scars in shallow waters; recovery of *Zostera marina* beds following blue mussel harvesting in Maine, US, has resulted in a mean recovery period of 9 – 11 years (Neckles *et al.*, 2005). MarLIN (2010) cites *Z. marina* to be moderately sensitive to the abrasion and physical disturbance pressure with a moderate score for recoverability.

17. Gill net, trammel nets and entangling nets on subtidal seagrass beds

17.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports primarily from the UK, Europe, and US. There is an evidence gap relating to the impacts of netting and static gears on seagrass in general.

17.2. Background

Fishing gears are commonly used where seagrass is found either in the form of bottom set gill or trammel nets. Impacts may occur through crushing by weights and anchors as part of netting gears or from footropes damaging or removing plant meristems and leaves (d'Avack *et al.*, 2014; Walmesmley *et al.*, 2015; Sørensen *et al.*, 2015). Whilst the potential for damage is lower per unit deployment compared to towed gear there is a risk of cumulative damage if use is intensive (Sørensen *et al.*, 2015). The severity of impacts may depend on whether disturbance occurs above (to leaves and stems) or below the ground (to roots/rhizomes) (Stephan *et al.*, 2000). Whilst there is potential for below-ground disturbance from netting anchors, there is no robust evidence to suggest that mechanical disturbance caused by fishing (or other) activity is beneficial, or that uprooted marine plants are likely to re-root themselves (Stephan *et al.*, 2000).

Seagrass meadows provide complex, supporting habitats for many commercial fish and invertebrate species such as oligochaetes, polychaetes, bivalves, and amphipods (Nordlund *et al.*, 2018; Duffy, 2006; Collins *et al.*, 2010). In a study of Mediterranean and Atlantic *Zostera spp.* meadows, Mosbahi *et al.* (2017) recorded a total of 232 taxa, with annelids and molluscs the dominant groups at study sites. They also provide nursery areas and have positive spill-over effects to adjacent habitats. In themselves subtidal seagrass beds provide shallow-water fishing grounds, though due to limitations in reporting information, the extent of exploitation and the impacts from the gear types deployed are not fully understood (Nordlund *et al.* 2018). Most seagrasses globally are declining and under threat from anthropogenic pressures (including from gill net fisheries), and indicators of seagrass health and resilience may help prevent further declines (Phair *et al.*, 2020).

Nordlund *et al.* (2018) undertook a global expert survey to understand the breadth and significance of fishing activity on seagrass habitats and found seagrass-based fisheries to be important and persistent at a global scale wherever seagrass exists. A wide range of fishing gears are used on seagrass generally driven by depth of water and the often-multi-species, nature of targeted fisheries, which may also include associated invertebrates. In developed countries, seagrass fisheries are target species specific; in the UK this fishery

consists of finfish, crustaceans and invertebrates via gill netting, hand netting, potting, rod/line, and hook/line activity (Nordlund *et al.*, 2018).

Targeted management to support the continued viability of seagrass meadows as resource providers is needed (Nordlund *et al.*, 2018); their report also provides direct comparisons of seagrass fishery gear types (including gill nets, entangling nets, and traps as per FAO: Fisheries & Aquaculture). This provides a common interpretation of fishing gears between countries and regions so that gear types and associated fisheries impacts may be compared at a wider scale to assess and address declines in seagrass extent.

17.3. Impacts

17.3.1. Abrasion (to the physical habitat and damage to epifauna)

Walmesley *et al.* (2015) and d'Avack *et al.* (2014) suggest that net gears (including weights, lines, and anchors) which are consistently set and hauled provide potential for abrasive movement and subsequent, considerable damage to seagrass meadows and their associated invertebrate fauna. This may present as leaf-shearing, meristem damage, uprooting, smothering, light reduction, and attenuation on species such as common eelgrass (*Zostera marina*) (Roberts *et al.*, 2010; Stephan *et al.*, 2000; Jenkin *et al.*, 2017). While the individual plant cannot regrow the lost leaf or stem, it may produce new growth and non-UK species such as *Posidonia*, are likely more susceptible to above-ground impacts (Stephan *et al.*, 2000) and potentially, seagrass wasting disease. Seed or flower shearing may be caused primarily by mobile gears or weights and anchors on static gears, impacting the plant's reproductive ability. Loss of flowers or seeds will affect the next year's growth of new plants and impact would likely be most detrimental for species that rely heavily on sexual reproduction (e.g., *Z. marina*) (Stephan *et al.*, 2000).

The nature of uprooting of plants (random and occasional) through snagging and displacement, and uprooting without sediment disturbance, may be caused by netting activity and ground contacting netting gears which may result in impacting seagrasses above-ground (Stephan *et al.*, 2000).

Long set times or ghost fishing from lost nets may prolong these impacts or displace more fragile seagrass species (NOAA, 1996) though no primary evidence is provided. Several factors may combine to affect the degree of damage from netting, such as the intensity of netting activity (number of nets), set time and set/haul frequency (JNCC and NE, 2011) with the potential for storms to exacerbate these impacts through increased net movement; seasonality and intensity of activity during sensitive spawning periods may also have impacts on structure, function, and recruitment (Stephan *et al.*, 2000). Sufficiently large numbers of nets, or cumulative use in an area could result in elevated levels of losses of submerged aquatic vegetation such as seagrass (Stephan *et al.*, 2000; Roberts *et al.*, 2010).

17.3.2. Penetration and/or disturbance of the substratum below the surface of the seabed, including abrasion

Netting gear anchors have below-ground impacts as they penetrate the sediment as described in d'Avack *et al.* (2014). The indirect impacts of this are disturbance and suspension of sediments into the water column and consequential siltation onto the seagrass leaves, potentially impairing photosynthetic capability, and growth (Collins *et al.*, 2010). Collins *et al.* (2010) describe such analogous processes from boat anchorage in Studland Bay, which exposes the underlying rhizomes with the potential for exacerbation of this from burrowing shore crabs. Sørensen *et al.* (2015) also describe the impacts of penetrative pressures on eelgrass in Denmark noting that floral density affects the extent of the removal when hauling gill net anchors. Four pronged, 30cm (prong length) anchors were noted to rest on the sandy substrates or penetrate the sand to a small degree; occasionally two of the anchor prongs were observed to be more than half buried. This study showed minor (a few tufts of algae and some eelgrass blades) or intermediate (up to 0.5 Kg of algae and many blades of eelgrass), removal of eelgrass in 10 of 15 (2014) and 8 of 10 (2015) anchor-haul observations. A steeper angle of anchor-haul results in a ploughing effect causing benthic structural changes, whereas the greater the distance between vessel and anchor also resulted in variations in impact distance. (Sørensen *et al.*, 2015).

17.3.3. Removal of species

d'Avack *et al.* (2014) states that this pressure is generally exerted by targeted fisheries for species associated with seagrass meadows (i.e., mussels, clams, and scallops), but that these species are fished by trawling, dredging, digging and raking. It is acknowledged, however, that seagrass plants and the sediment they grow in may be directly removed or damaged by static gears targeting other species. Any incidental removal of seagrass or associated species may alter the character and structure of the biotope with consequential change in overall species richness and ecosystem services provision (d'Avack *et al.*, 2014). Removal of incidental algae and eelgrass has also been recorded (Sørensen *et al.*, 2015).

17.3.4. Recovery of Habitat and Communities

Duffy (2006) asserts that genetic diversity enhances seagrass growth and resistance to disturbance. Dominant, supporting species such, as seagrasses, have bottom-up influences on associated or dependent communities, therefore, such species can influence the structure, functioning and recovery of entire ecosystems (Whitham *et al.*, 2003; Reusch and Hughes 2006). Concomitant abrasion and crushing pressure may also intensify impacts and increase the risk of cumulative impacts over time with the risks most likely to occur from potting and netting activity (via weights and anchors) leading to an extended recovery period, though not quantified (d'Avack *et al.*, 2014).

18. Pots/creels (crustacea/gastropods) on Ross worm (*Sabellaria spinulosa*) reefs and subtidal biogenic reefs: *Sabellaria* spp

18.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports primarily from the UK. There is an evidence gap relating to the impacts of netting and static gears on *Sabellaria* spp. reefs in general.

18.2. Background

Sabellaria spinulosa reef biotopes are found on a range of substratum types including rock and mixed sediments (Gibb *et al.*, 2014).

Walmsley *et al.* (2015) state, that at time of writing, there was no primary evidence for the impacts of potting activity on *S. spinulosa* reefs. Key evidence gaps are around potting intensity and location, long time series data and natural variability (Walmsley *et al.*, 2015). While there has been more recent interest, and sensitivity assessments exist (Hall *et al.*, 2008), these evidence gaps persist (Tillin *et al.*, 2010; Gibb *et al.*, 2014) and further research is recommended.

Sabellaria spp. reefs may be impacted by both static and towed gear types (Roberts *et al.*, 2010) depending on location of the reef and exposure to various pressures, e.g., a subtidal reef is unlikely to be exposed to trampling, however at spring tides this may occur if the reef is exposed. Gibb *et al.* (2014) attribute a higher sensitivity to pressures to *S. spinulosa*; *S. alveolata* is considered more robust due to its exposure to pressures associated with intertidal activity, though not considered in this review.

Existing sensitivity assessments are based on trawling or dredging activity (Walmsley *et al.*, 2015; d'Avack *et al.*, 2014). It is assessed as having a medium sensitivity to all levels of potting, due to its robust nature and ability to recover quickly (Walmsley *et al.*, 2015; d'Avack *et al.*, 2014).

JNCC and NE (2011) agreed that the sensitivity of *Sabellaria* spp. reefs to static gears is low to medium depending on intensity, while other researchers have considered sensitivity as minimal (Holt *et al.* (1998). Tillin *et al.* (2010) considered *S. alveolata* and *S. spinulosa* reefs to have a low sensitivity to surface abrasion from static fishing gear types. The intertidal versus subtidal occurrence of *Sabellaria* species have different exposure to different pressures and may be dependent on bed location (which may contribute to ephemerality of *S. alveolata* reefs [Oliver, 2016]) and the phase of the tidal cycle (i.e., spring tides) may expose *S. spinulosa* beds at low tide. Walmsley *et al.*, (2015) concur, stating that evidence from one area may not be directly applicable due to site-specific

differences, though the preferred settlement substrata include coarse and mixed sediments (MarLIN, 2006).

18.3. Impacts

18.3.1. Abrasion (to the physical habitat and damage to epifauna)

Potting on *Sabellaria* spp. reefs is generally considered to have a low impact due to reefs robust structure coupled with low intensity and frequency of potting and the small footprint of the activity (Walmesley *et al.*, 2015). This is a relative statement grounded on trawling-based sensitivity assessments and a broad consensus regarding sensitivity of *S. spinulosa*. Trawling also exerts a penetrative pressure (Gibb *et al.*, 2014) which is much more destructive to reef structures.

The mechanisms of abrasion pressure derive from the pots themselves which may have different construction materials and differ in dimensions and weight. As with potting on rock, the pots may abrade the surface of *S. spinulosa* reefs during shooting and settlement and during hauling where they may be dragged for some distance (Hartnoll, 1998; Eno *et al.*, 2001; Hall *et al.*, 2008; Rees, 2018; Johnson, 2002). Storm conditions may also contribute to abrasion as the pots, ropes, and weights may move in strong currents.

Sensitivity to this abrasion pressure can modify the bioengineering *Sabellaria* and associated communities (Salomidi *et al.*, 2012) predominantly, through high impact trawling activities (Bouma *et al.*, 2009), but also through potting to a lesser degree. Any loss of reef habitat structure from abrasion can drive reduced abundance, biomass and species richness and consequential ecosystem functioning (Salomidi *et al.*, 2012). *Sabellaria*'s high resilience makes it less sensitive to abrasion (Tillin *et al.*, 2010). Abiotic factors (e.g., current strength, sediment supply) in the local environment can compound impacts from potting which may also contribute to natural variability (Salomidi *et al.*, 2012).

Gibb *et al.* (2014) state that abrasion at the surface of reefs is likely to damage the ends of the worm tubes and may cause greater damage where areas are broken apart. Sewell and Hiscock (2005) suggest there is little likelihood of damage to *S. spinulosa* reefs from potting apart from bycatch and abandoned, lost, or discarded pots abrading reefs or crusts.

18.3.2. Removal of species

The removal of species (e.g., juvenile flatfish that) preferentially prey on *S. spinulosa* could be beneficial to this species and predation could be reduced, however there is no empirical evidence or relationship information to confirm this (Gibb *et al.*, 2014),

Gibb *et al.*, (2014) cite previous studies which show a predator-prey relationship between *S. spinulosa* and non-commercial species butterflyfish (*Pholis gunnellus*) and dragonet (*Callionymus lyra*). *Carcinus maenas* are also known to predate *S. spinulosa* (Taylor 1962; Bamber & Irving 1997 in Gibb *et al.*, 2014). The brittle star, *Ophiothrix fragilis* which can

form dense aggregations also compete with *S. spinulosa* for food and space and removal could benefit the wider reef, *S. spinulosa* recruitment and epifaunal species. While *S. spinulosa* is considered not to be sensitive to removal of target and non-target species, this demonstrates the principle that target and non-target species have the potential to affect *Sabellaria* larval settlement and survival, and subsequent reef establishment and maintenance.

18.3.3. Recovery of Habitat and Communities

Following disturbance, fracturing damage, or removal the reef structure itself may not disappear as its recovery capacity means damaged parts of the reef can be rebuilt within a few days (Salomidi *et al.*, 2012) depending on the extent and nature of the damage. Recovery is dependent on the supply of suitable material with which to repair the damaged part of the tube and a lack of material could result in erosion of the reef (Last *et al.*, 2011). Reef associated fauna are subjected to an immediate impact which could have longer term impacts on the communities' recovery (Salomidi *et al.*, 2012).

Some evidence relating to trampling on intertidal *S. alveolata* reefs has been gathered (the reef taking 23 days to recover from severe damage), although no specific recovery evidence exists for subtidal potting activity, however, there remains significant information gaps regarding recovery rates, stability, and persistence of *S. spinulosa* reefs (Gibb *et al.*, 2014).

Extrapolating results of (post-beam trawl) recovery studies from *S. alveolata* to *S. spinulosa* suggests that, if sufficient proportions of reef and worms survive, recovery rate through repair and larval colonisation could occur within 2-10 years (Gibb *et al.*, 2014). It is anticipated that abrasion pressure from potting activities would not be so severe, and recovery would take significantly less time, though no empirical evidence for this was found.

19. Gill nets, trammel nets and entangling nets on Ross worm (*Sabellaria spinulosa*) reefs and subtidal biogenic reefs: *Sabellaria spp*

19.1. Evidence

This review is based on information sourced from peer-reviewed scientific journals and research reports primarily from the UK. There is an evidence gap relating to the impacts of netting and static gears on *Sabellaria* reefs in general.

19.2. Background

Impacts from static netting gears on *Sabellaria spinulosa* reefs are considered comparatively low - medium, though intensity and variation in local environmental conditions may differ (Tillin *et al.*, 2010; Gibb *et al.*, 2014; Sewell and Hiscock 2005; Salomidi *et al.*, 2012, Walmesley *et al.*, 2015). There is acknowledgement that *Sabellaria* reefs may be impacted by both static and towed gear types (Roberts *et al.*, 2010). Further, Sewell and Hiscock (2005) suggest impacts to *S. spinulosa* reefs from netting are most likely to originate from bycatch and ghost fishing (from abandoned, lost, or discarded weights and anchors abrading reefs or crusts).

Walmesley *et al.*, (2015) state that at time of writing, there was no primary evidence for the impacts of netting activity on *S. spinulosa* reefs. While there has been more recent interest and sensitivity assessments exist (Hall *et al.*, 2008), there is an acknowledged lack of evidence for netting on *S. spinulosa* reef (Tillin *et al.*, 2010). Several evidence gaps exist for this feature, primarily around substrate, fishing intensity, long time series data and natural variability (Walmesley *et al.*, 2015) and further research is widely recommended.

The sensitivity assessments made are based on trawling or dredging activity (Walmesley *et al.*, 2015; d'Avack *et al.*, 2014). *S. spinulosa* is assessed as having a medium sensitivity to netting gear, due to its robust nature and rapid recovery period (Walmesley *et al.*, 2015; d'Avack *et al.*, 2014).

JNCC and NE (2011) agreed that the sensitivity of *Sabellaria* reefs to static gears is low to medium depending on intensity, while other researchers have considered sensitivity as minimal (Holt *et al.*, 1998) and low; Tillin *et al.* (2010) considered *S. spinulosa* reefs to have a low sensitivity to surface abrasion from static fishing gear types.

19.3. Impacts

19.3.1. Abrasion (to the physical habitat and damage to epifauna)

Netting on *S. spinulosa* reefs is generally considered to have a low impact due to reefs robust structure coupled with low intensity and frequency of netting and the small footprint of the activity (Walmesley *et al.*, 2015). This is a relative statement grounded on trawling-based sensitivity assessments and a broad consensus regarding sensitivity of *S. spinulosa*. The anchors used with static nets may exert a penetrative pressure as the anchor grabs hold, though no direct evidence was found for this. Trawling exerts a penetrative pressure (Gibb *et al.*, 2014) which may be destructive to reef structures.

The mechanisms of abrasion pressure derive from the nets themselves, weights, lines and anchors. As with netting on rock, elements of the net gear may abrade the surface of *S. spinulosa* reefs during deployment and hauling where weighted lines or anchors may be dragged for some distance (Hartnoll, 1998; Eno *et al.*, 2001; Hall *et al.*, 2008; Rees, 2018; Johnson, 2002). Storm conditions may also contribute to abrasion as the weights and anchors may move in strong currents.

Sensitivity to this abrasion pressure can modify *S. spinulosa* reefs and associated communities (Salomidi *et al.*, 2012) predominantly, through high impact trawling activities (Bouma *et al.*, 2009) but also through the action of netting anchors. Any loss of reef habitat structure from abrasion from weights or associated damage from anchor setting, can drive reduced abundance, biomass and species richness and consequential ecosystem functioning (Salomidi *et al.*, 2012). *Sabellaria*'s high resilience makes it less sensitive to abrasion (Tillin *et al.*, 2010). Abiotic factors (e.g., current strength, sediment supply) in the local environment can compound impacts from netting activity which may also contribute to natural variability (Salomidi *et al.*, 2012).

Gibb *et al.* (2014) state that abrasion at the surface of reefs is likely to damage the ends of the worm tubes and may cause greater damage where areas are broken apart (via net anchors).

19.3.2. Removal of species

Gibb *et al.* (2014) report evidence of potential effects of the removal of other species on the *S. spinulosa* reefs, however no evidence of significant biological effects from the removal of non-target species associated with these reefs was identified.

The removal of species (e.g., juvenile flatfish) that preferentially prey on *S. spinulosa* could be beneficial to this species and predation could be reduced, however there is no empirical evidence or relationship information to confirm this (Gibb *et al.*, 2014).

Gibb *et al.* (2014) also cite previous studies which show a predator-prey relationship between *S. spinulosa* and non-commercial species butterfish (*Pholis gunnellus*) and dragonet (*Callionymus lyra*). *Carcinus maenas* are also known to predate *S. spinulosa* (Taylor 1962; Bamber & Irving 1997 in Gibb *et al.*, 2014). The brittle star, *Ophiothrix fragilis* which can form dense aggregations also compete with *S. spinulosa* for food and space and removal could benefit the reef itself, *S. spinulosa* recruitment and epifaunal species.

In Isle of Man waters, *S. spinulosa* occurs over fine sand, in high abundance and with associated surface-dwelling organisms e.g., dead man's fingers (*Alcyonium digitatum*) (Hinz *et al.*, 2009). The weights and anchor elements of netting gears may exert crushing pressures on surface-dwelling organisms or detach them from the *Sabellaria* reef (as with a rock substrate) though no direct evidence of this was found.

More generally, there is a lack of evidence of direct impacts from removal of both species associated with *S. spinulosa* reefs from netting activity.

19.3.3. Recovery of Habitat and Communities

Following disturbance, fracturing damage, or removal the reef structure itself may not disappear as its recovery capacity means damaged parts of the reef can be rebuilt within a few days (Salomidi *et al.*, 2012) depending on the extent and nature of the damage. While there is no evidence of significant structural impacts from static nets, Vorberg (2000) (*in* Gibb *et al.*, 2014) reported post-trawling impacts *S. alveolata* reefs appeared repaired

within four to five days. The daily growth rate of the worm tubes during a restoration phase was significantly higher than undisturbed growth (undisturbed: 0.7mm, after removal of 2cm of surface: 4.4mm). Recovery of thin encrusting reefs (less than 2cm) may therefore be relatively rapid (Gibb *et al.*, 2014). Reef associated fauna, however, are subjected to an immediate impact (Rabaut *et al.*, 2008 – in Salomidi *et al.*, 2012). As the impact from netting activity is considered low along with sensitivity to the pressure, it is unlikely that significant long-term impacts or loss of reef may be attributable solely to this gear type.

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