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Management
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Stage 3 Fishing Gear MPA Impacts Evidence: Traps



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Stage 3 Fishing Gear MPA Impacts Evidence: Traps

Contents

Executive Summary	4
1 Introduction.....	5
1.1 Key definitions	5
1.2 Structure of this document.....	6
2 Overview of gear group: traps.....	6
3 MPA features.....	7
4 Sea-pen and burrowing megafauna communities	8
4.1 Overview of the sensitivity of sea-pen and burrowing megafauna communities to traps	9
4.1.1 Sensitivity – resistance to damage	9
4.1.2 Recovery – rate of recovery.....	10
4.2 Level of literature, caveats and assumptions.....	11
4.3 The pressures of traps on sea-pen and burrowing megafauna communities ..	12
4.3.1 Abrasion or disturbance of the substrate on the surface of the seabed	12
4.3.2 Removal of target species	13
4.3.3 Removal of non-target species	14
4.4 Variation in impacts	15
4.5 Summary of the effects of traps on sea-pen and burrowing megafauna communities	16
5 Fan mussel.....	17
5.1 Overview of the sensitivity of fan mussel to traps.....	18
5.1.1 Sensitivity – resistance to damage	18
5.1.2 Recovery – rate of recovery.....	18
5.2 Level of literature, caveats and assumptions.....	19
5.3 The pressures of traps on fan mussel	19
5.3.1 Abrasion or disturbance of the substrate on the surface of the seabed and penetration and/or disturbance of the substratum below the surface of the seabed, including abrasion	20

5.3.2	Removal of non-target species	20
5.4	Variation in impacts	20
5.5	Summary of the effects of traps on fan mussel.....	21
6	Ocean quahog.....	22
6.1	Overview of the sensitivity of ocean quahog to traps.....	22
6.1.1	Sensitivity – resistance to damage	22
6.1.2	Recovery – rate of recovery.....	23
6.2	Level of literature, caveats and assumptions.....	24
6.3	The pressures of traps on ocean quahog	24
6.3.1	Abrasion or disturbance of the substrate on the surface of the seabed	24
6.4	Variation in impacts	25
6.5	Summary of the effects of traps on ocean quahog	25
7	Rocky reef	26
7.1	Overview of the sensitivity of rocky reef to traps.....	28
7.1.1	Sensitivity – resistance to damage	28
7.1.2	Recovery – rate of recovery.....	29
7.2	Level of literature, caveats and assumptions.....	30
7.3	The pressures of traps on rocky reef	31
7.3.1	Abrasion or disturbance of the substrate on the surface of the seabed	31
7.3.2	Removal of target species	33
7.3.3	Removal of non-target species	34
7.4	Variation in impacts	35
7.5	Summary of the effects of traps on rocky reef	37
8	Biogenic reef (<i>Sabellaria spp.</i>)	38
8.1	Overview of the sensitivity of biogenic reef to traps.....	38
8.1.1	Sensitivity – resistance to damage	38
8.1.2	Recovery – rate of recovery.....	39
8.2	Level of literature, caveats and assumptions.....	39
8.3	The pressures of traps on biogenic reef	40
8.3.1	Abrasion/disturbance of the substrate on the surface of the seabed	41
8.3.2	Removal of non-target species	42

8.4	Variation in impacts	43
8.5	Summary of the effects of traps on biogenic reef	43
9	Annex I sandbanks which are slightly covered by sea water all the time and MCZ subtidal sediment habitats.....	44
9.1	Feature summaries.....	47
9.1.1	Sandbanks.....	47
9.1.2	Supporting habitats.....	48
9.2	Overview of the sensitivity of sandbanks and sediments to traps.....	48
9.2.1	Sensitivity – resistance to damage	48
9.2.2	Recovery – rate of recovery.....	49
9.3	Level of literature, caveats and assumptions.....	49
9.4	The pressures of traps on sandbanks and sediments	50
9.4.1	Abrasion/disturbance of the substrate on the surface of the seabed	50
9.4.2	Removal of target species	51
9.4.3	Removal of non-target species	51
9.5	Variation in impacts	51
9.6	Summary of the effects of traps on sandbanks and sediments	53
10	References	54
	Annex 1 Gear pressures on sensitive features - traps.....	69

Executive Summary

This document collates and analyses the best available evidence on the impacts of traps on MPA features and will inform site level assessments of the impact of traps on MPAs as part of Stage 3 of the MMO's work to manage fishing in MPAs.

Traps have the potential to impact some MPA features, therefore management of these fishing gears could be required. For each MPA, a site level assessment considering the site conservation objectives, intensity of fishing activity taking place and exposure to natural disturbance will be completed to determine whether management will be required.

1 Introduction

The Marine Management Organisation (MMO) is the principal regulator for England's seas, including leading the assessment and management of fishing for marine protected areas (MPAs) offshore of 6 nautical miles (nm)¹.

This document forms part of MMO's Stage 3 work to achieve the government's aim of having appropriate fisheries management measures in place for all offshore MPAs in English waters by the end of 2024. It is one of a suite of documents which focus on the interaction of fishing gear on particular designated features, and it will support the delivery of site level assessments.

This document describes the impact of traps on protected habitats and species (i.e. designated features). It describes the potential for pressures and impacts caused by traps on designated features within MPAs by gathering and analysing the available evidence for gear-feature interactions.

The [Stage 3 Call for Evidence Introduction](#) provides further background information and details of other documents produced.

1.1 Key definitions

A separate glossary in the Stage 3 Call for Evidence Introduction includes the important terms used in this document. Wherever possible these are taken from [Natural England's Glossary of terms used within conservation advice packages \(CAPs\)](#).

The following terms are particularly important when reading this document and are described further in Figure 1.

Designated Feature ('feature') - A species, habitat, geological or geomorphological entity for which an MPA is identified and managed.

Sensitivity – The sensitivity of a feature (species or habitat) is a measure that is dependent on the ability of the feature (species or habitat) to resist change and its ability (time taken) to recover from change.

Pressure - the mechanisms through which an activity has an effect on a feature.

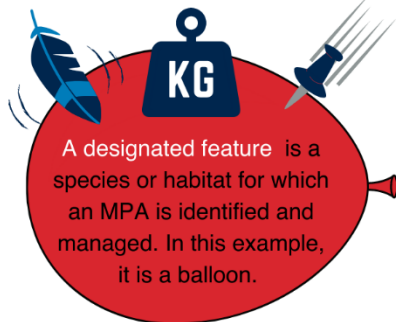
Impact - the consequence of pressures (such as habitat degradation) where a change occurs that is different to that expected under natural conditions.

¹ Inshore fisheries and conservation authorities (IFCAs) are responsible for managing fishing within 6 nm.

The sensitivity of MPA features to pressures

To understand how different pressures from activities, such as fishing, might affect the designated features of an MPA, MMO look at the available evidence on activities, features and their sensitivities to pressures.

Pressures are the mechanisms through which an activity has an effect on a feature. In this example the activities are a weight, a feather or a pin, and the pressures are pressing, brushing or piercing.



The sensitivity of a feature is a measure that is dependent on the ability of the feature to resist change and its ability (time taken) to recover from change. In this example, a balloon can resist being brushed by a feather or pressed upon by a light weight.

An impact is the consequence of a pressure, where a change occurs in the species or habitat that is different to what would be expected naturally." In this example, this could be a compressed or popped balloon.

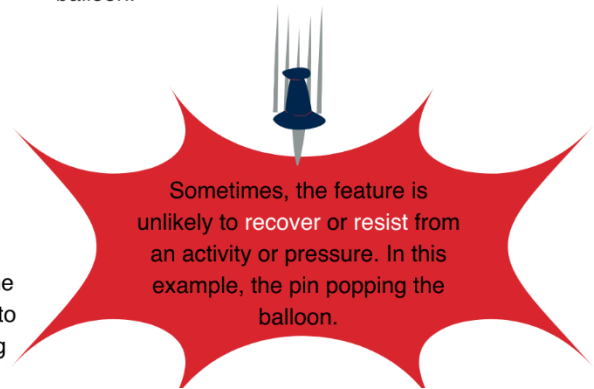


Figure 1. The sensitivity of MPA features to pressures.

1.2 Structure of this document

Section 2 describes the types of fishing gears considered in this document.

Section 3 lists the MPA features considered and references the evidence sources used in this document.

Sections 4 to 9 describe the available evidence of the pressures resulting from the fishing gears on different MPA features. Each section also describes evidence about the sensitivity of each feature to damage and how resilient it is (how quickly a feature can recover).

Annex 1 Gear pressures on sensitive features - traps lists pressures which are common to all features. Any feature-specific pressures with insufficient evidence are listed in the relevant section.

2 Overview of gear group: traps

Traps are stationary structures of many shapes and sizes into which fish and shellfish are drawn by bait or other attractants (He et al., 2021). A pot is a kind of trap, usually set on the sea floor, with a small enclosure that attracts species through one or more entrances allowing their entry but preventing or hindering their escape (He et al., 2021). The term 'trap' is used interchangeably with pot in the literature and by the fishing industry in many fisheries and in many locations. Smaller pots are also called 'creels' (He et al., 2021). Pots are probably used over sandy habitats to a lesser degree than in rocky habitats (Roberts et al., 2010).

There may be site level instances where litter derived from fishing vessels or ghost gears have an impact, however, this pressure is not appropriate to manage in a localised way at MPA level for fisheries only. International legislation is in place, such as Annex V of the International Convention for the Prevention of Pollution from Ships (1973) (International Maritime Organization, 2019). This pressure will therefore not be covered in this review. Further information about different traps can be found at: www.fao.org/3/cb4966en/cb4966en.pdf.

3 MPA features

This section identifies features which have been identified as potentially sensitive to traps. Table 1. Feature descriptions references out to descriptions of the features from a recognised source. These sensitivities were derived using advice from the Joint Nature Conservation Committee (JNCC) and Natural England (NE) and review of the available scientific literature. Please see Annex 1 for a summary of the pressures of traps on the features described in this document and their associated sensitivities.

Table 1. Feature descriptions

Feature name	Feature Description
Sea-pen and burrowing megafauna communities	JNCC: Seapens and burrowing megafauna in circalittoral fine mud MarLIN: Seapens and burrowing megafauna in circalittoral fine mud
Fan mussel	MarLIN: Fan mussel (<i>Atrina fragilis</i>)
Ocean quahog	MarLIN: Ocean quahog (<i>Arctica islandica</i>)
Rocky reef	EUNIS: Atlantic and Mediterranean moderate energy circalittoral rock EUNIS: Atlantic and Mediterranean high energy circalittoral rock EUNIS: Circalittoral rock and other hard substrata JNCC: Annex I reef JNCC: Circalittoral rock (and other hard substrata) JNCC: High energy circalittoral rock JNCC: Moderate energy circalittoral rock
Biogenic reef (<i>Sabellaria spp.</i>)	JNCC: Annex I reef JNCC: Reefs MarLIN: Ross worm (<i>Sabellaria spinulosa</i>) MarLIN: Honeycomb worm (<i>Sabellaria alveolata</i>) OSPAR Commission: <i>Sabellaria spinulosa</i> reefs

Feature name	Feature Description
Annex I sandbanks ² and MCZ sediments ³	EUNIS: Subtidal coarse sediment EUNIS: Subtidal sand EUNIS: Subtidal mud EUNIS: Subtidal mixed sediment JNCC: Sandbanks which are slightly covered by sea water all the time

This document focusses on traps. Annex 1 contains tables summarising which features are affected by traps. Annex 1 shows that not all features are sensitive to all types of pressures from traps. Where a feature is potentially sensitive to traps (based on its resilience to the pressure and ability to recover) the interaction is considered in sections 4 to 9 below. Each section lists the relevant pressures to which the features are sensitive. It also lists those pressures where insufficient evidence has been found to indicate whether it is sensitive/not sensitive.

4 Sea-pen and burrowing megafauna communities

This section brings together and analyses the available evidence on how traps affect sea-pen and burrowing megafauna communities.

Sea-pen and burrowing megafauna communities have been identified by OSPAR as a habitat of key conservation importance as defined under Annex V of the 1992 OSPAR Convention (OSPAR, 1992; OSPAR Commission, 2010) and are protected in UK waters by various legislation. They are a designated feature of the following offshore marine conservation zones (MCZs): East of Haig Fras (JNCC, 2021d), Farnes East (JNCC, 2017a), Greater Haig Fras (JNCC, 2018c), North West of Jones Bank (JNCC, 2018g) and West of Walney (JNCC, 2018k; Natural England and JNCC, 2018).

The habitat is defined using the OSPAR definition (OSPAR Commission, 2021): 'Plains of fine mud, at water depths ranging from 15 to 200 m or more, which are heavily bioturbated by burrowing megafauna with burrows and mounds typically forming a prominent feature of the sediment surface'. The habitat may include conspicuous populations of sea-pens, typically *Virgularia mirabilis* and *Pennatula phosphorea*. The burrowing crustaceans present may include *Nephrops norvegicus*, *Calocaris macandreae* or *Callianassa subterranea*. In the deeper fjordic lochs which are protected by an entrance sill, the tall sea-pen *Funiculina quadrangularis* may also be present. The burrowing activity of megafauna creates a complex habitat, providing deep oxygen penetration. This habitat occurs extensively in sheltered

² Annex I Sandbanks which are slightly covered by sea water all the time

³ Marine conservation zone subtidal sediment habitats include: subtidal coarse sediment, subtidal sand, subtidal mixed sediments, subtidal mud.

basins of fjords, sea lochs, voes and in deeper offshore waters such as the North Sea and Irish Sea basins.

Although they occur in the same muddy habitats, sea-pen and burrowing megafauna communities are functionally and ecologically different and are not necessarily associated with one another (Hill et al., 2020). Sites with this feature may have an abundance of burrowing megafauna but lack sea-pens (Hill et al., 2020). It is possible that this may be due to environmental factors or because of human pressures. Some forms of sampling may fail to indicate the presence of sea-pens where they have been visually recorded via other methods, so it could be possible that sea-pens occur more frequently than research suggests (Hill et al., 2020). There is no single keystone species essential to the feature or the community (Hill et al., 2020), but burrowing megafauna are an essential element of the habitat.

N. norvegicus can be fished either by trawl or by static gear such as traps or creels. It has been generally thought that creel fishing causes minimal damage to sea-pen biotopes (Eno et al., 1996; Hughes, 1998).

The evidence base for all relevant gear interactions with this feature is not extensive and uncertainty exists around its sensitivity to fisheries impacts.

4.1 Overview of the sensitivity of sea-pen and burrowing megafauna communities to traps

4.1.1 Sensitivity – resistance to damage

This feature is considered highly vulnerable to physical disturbance to the seabed or mechanical damage from demersal fishing gear because the gear has the potential to damage the feature's fragile components such as sea-pens, and can change benthic community structure and function, and resuspend sediment particles (OSPAR Commission, 2010; Gonzalez-Mirelis and Buhl-Mortensen, 2015).

Dinmore et al. (2003) stated that large, slow growing species such as sea-pens are particularly vulnerable to trawling. Sea-pens are more sensitive to removal by penetrative gear, as it can entirely remove animals from their burrows (Hill et al., 2020). The Marine Life Information Network (MarLIN) has therefore assessed resistance as 'Low' for all three sea-pen species commonly found in this feature (*V. mirabilis*, *F. quadrangularis* and *P. phosphorea*) (Hill et al., 2020). For definitions of resistance (tolerance), resilience (recovery) and sensitivity rankings from the Marine Evidence based Sensitivity Assessment (MarESA) (Tyler-Walters et al., 2018), see the glossary in the [Stage 3 Call for Evidence Introduction](#).

Many species of sea-pens such as *V. mirabilis* and *P. phosphorea* can withdraw into tubes in the sediment (Hoare and Wilson, 1977; Ambroso et al., 2013). It has been hypothesised, therefore, that they may be able to avoid approaching demersal fishing gears (Hughes, 1998). It should be noted, however, that the penetration depths of demersal gears in mud habitats can vary from 3 to 6 cm (Gubbay and

Knapman, 1999), and for otter trawl doors from ≤ 15 to 35 cm (Eigaard et al., 2016). Also, sea-pen behavioural observations have only noted that individuals can withdraw completely below the sediment surface without specifying depth or speed. It is also unclear whether this withdrawal could be triggered by approaching gear as this behaviour is not well understood (Ambroso et al., 2013). Their withdrawal has been described as rhythmic and unsynchronised (Langton et al., 1990). Numerous studies also hypothesise that their ability to withdraw makes measuring sea-pen abundance extremely difficult (Birkeland, 1974; Eno et al., 2001; Greathead et al., 2007, 2011). It should be noted that the sea-pen *F. quadrangularis* cannot withdraw into the sediment (Hill et al., 2020).

Some species of burrowing megafauna may be able to avoid demersal fishing gears by burrowing beneath the sediment surface. For example, *N. norvegicus* form burrows in the sediment of 20 to 30 cm depth (Aguzzi and Sardà, 2008). Despite this ability, there is still a successful targeted fishery. This is because *N. norvegicus* is a burrowing crustacean with behavioural adaptations to ambient light (Ball et al., 2000). Burrow emergence is highest at dawn and dusk in shallower grounds, and gets closer to midday in deeper waters (Chapman, 1980). Fishing effort is targeted to exploit this behaviour, increase catch rates, and minimise gear avoidance. Generally, larger, slow-growing burrowing megafauna are more vulnerable to demersal fishing gear than smaller individuals that are pushed aside with fluidised sediments rather than damaged (Dinmore et al., 2003).

A review on the response of benthic fauna to experimental demersal fishing found that a gear pass reduced benthic invertebrate abundance by 26% and species richness by 19%, indicating that many species are sensitive (Sciberras et al., 2018). The United Nations General Assembly (United Nations General Assembly, 2006) defines sea-pen and burrowing megafauna communities as sensitive habitats that 'are easily adversely affected by human activity and/or if affected are expected only to recover over a very long period, or not at all'. The Sciberras review demonstrated that reductions in abundance and species richness were highly dependent on specific gear type, habitat type and the site's history of fishing disturbance. More penetrative gears, such as hydraulic dredges, had a significantly larger impact than those that penetrate less. Habitats with a higher percentage content of mud saw greater reductions in community abundance than those with lower mud content, and abundance also decreased more in historically undisturbed areas compared to previously disturbed areas (Sciberras et al., 2018).

4.1.2 Recovery – rate of recovery

Recovery from damaging activities will depend on the intensity and frequency of the impact and the recruitment processes of a species. Literature on the recruitment processes of sea-pens remains limited. Hughes, (1998) suggested that they are characterised by patchy recruitment, slow growth and long lifespans. Greathead et al. (2007) also described sea-pens as having a patchy site distribution likely related

to patchy larval settlement processes. Habitats formed by slow growing and long-lived specimens such as hydroids, corals (Troffe et al., 2005) or sea-pens are highly sensitive to pressures associated with fishing, suggesting that even with a reduced level of effort, fishing activity could cause considerable damage and prevent habitat recovery (Greathead et al., 2015).

Sites that are more intensely impacted (for example, through penetration and/or disturbance of the substratum below the surface of the seabed, including abrasion) or frequently disturbed are likely to take longer to recover than those with less damaging pressures (for example, abrasion or disturbance of the substrate on the surface of the seabed) or less disturbance.

The recovery rates of burrowing megafauna such as *N. norvegicus* will also depend on the spatial scale of impact and the recruitment processes of the species. Time to sexual maturity for *N. norvegicus* is 2.5 to 3 years and larval stages spend about 50 days as plankton, allowing for high potential dispersal (Hill et al., 2020). Post-settled individuals show limited migration capacity (Rice and Chapman, 1971) and are habitat limited due to their substrate requirements (Ungfors et al., 2013). This means that well-defined boundaries exist for *N. norvegicus* fisheries. The *N. norvegicus* component of the feature may therefore have a medium resilience to disturbance (likely recovering within 2 to 10 years, as defined by MarESA (Tyler-Walters et al., 2018)), depending on the scale of removal at each site (Hill et al., 2020).

Evidence from fishing grounds shows that populations of *N. norvegicus* can persist in areas where they are targeted for removal, suggesting a reasonable level of resilience against repeated disturbance. However due to a lack of historical population data it is unclear how much of the population is removed and therefore how populations would recover if disturbance was completely removed (Hill et al., 2020).

Sciberras et al., (2018) found that sessile and low mobility benthic fauna with longer lifespans took longer to recover after demersal fishing (> 3 years, categorised by MarESA as a medium recovery rate (Tyler-Walters et al., 2018)) than mobile species with shorter lifespans (< 1 year, categorised by MarESA as a high recovery rate (Tyler-Walters et al., 2018)). This is partly because mobile groups like polychaetes have high intrinsic rates of growth, but could also be because gastropod, malacostracan and ophiuroid species are able to migrate quickly and colonise areas.

4.2 Level of literature, caveats and assumptions

There is limited literature available on fishing gear interactions with sea-pen and burrowing megafauna communities, the majority of which concerns active fishing gears, and in the UK the research is primarily conducted in the Irish Sea. The use of pots or creels is generally considered far less damaging to benthic habitats than the use of mobile gears (Sewell and Hiscock, 2005). As such, less targeted research into

the potential impacts exists. The limited research concerning the potential impacts of traps, pots or creels on this feature is discussed here.

Sea-pen and burrowing megafauna communities tend to occur within sedimentary MCZ features. This is the case for all the MCZs listed in Section 4. They typically inhabit mud biotopes that fall under European University Information Systems' (EUNIS) habitat A5.3: sublittoral mud (EEA, 2019c; Hill et al., 2020).

4.3 The pressures of traps on sea-pen and burrowing megafauna communities

As a result of the use of traps, this feature may be sensitive to the following pressures, so they are considered in this document:

- abrasion or disturbance of the substrate on the surface of the seabed
- removal of target species (this is not classed as sensitive, but we have been advised to consider it in this instance).
- removal of non-target species

There is insufficient evidence available to determine whether this feature is sensitive to the following pressures as a result of the use of traps:

- hydrocarbon and polycyclic aromatic hydrocarbon (PAH) contamination
- introduction or spread of invasive non-indigenous species
- litter
- synthetic compound contamination
- transition elements and organo-metal contamination.

4.3.1 Abrasion or disturbance of the substrate on the surface of the seabed

Abrasion and penetration impacts from traps are possible through the interaction between the seabed and the gear itself, including associated lines and anchors. Anchoring can cause physical damage to static megafaunal species such as sea-pens (Hughes, 1998). Damage is most likely to occur if the gear moves across the seabed during hauling of gear or when the gear is subject to strong tides, currents or storm activity. Traps do not penetrate the surface of the seabed so burrowing megafauna is unlikely to be affected.

There is limited direct evidence of the impacts of static gears such as traps on the physical environment that sea-pen and burrowing megafauna communities inhabit. There is, however, potential for impacts to the biological communities which are discussed below.

Experimental studies show that, overall, surface abrasion by pots and creels is unlikely to affect sea-pens adversely (Kinnear et al., 1996; Eno et al., 2001). Eno et al. (2001) used SCUBA divers and video analysis to conduct surveys on the effects of the use of *Nephrops* creels on sea-pen species in Scottish lochs. The study

assessed the recovery and survival of sea-pen species after dragging, uprooting and smothering by creels for a maximum period of six days after the disturbance. It was noted that creels consistently hit the seabed at an angle, with the bottom edge of the door striking first. Video footage showed that the creels created a pressure wave that was able to bend the sea-pens away from the creel just before contact.

The study found that *F. quadrangularis* and *P. phosphorea* were able to right themselves when dislodged by traps, with all *P. phosphorea* individuals re-established and 50% of *F. quadrangularis* re-established after 72 hours (Eno et al., 2001). *V. mirabilis* was found to withdraw into its burrow rapidly (ca 30 seconds) and could not be uprooted by dragged creels (Hoare and Wilson, 1977; Eno et al., 2001; Ambroso et al., 2013). All three sea-pen species were found to recover to an upright position within 24 to 72 hours from the effects of being dragged over by pots or creels with the exception of one individual *F. quadrangularis*. (Eno et al., 2001).

Additionally, the slow-growing, long-lived, pink sea fan *Eunicella verrucosa* was found to flex under the weight of pots as they passed and then return back to an upright position (Eno et al., 2001) further indicating that erect, sessile organisms such as sea-pens are relatively insensitive to the physical impacts of potting.

Experimental potting studies found that both *P. phosphorea* and *F. quadrangularis* were laid flat before contact as a result of the pressure wave caused by approaching pots/creels (Kinnear et al., 1996). The authors also noted that *P. phosphorea* and *F. quadrangularis* were occasionally removed from the substratum by creels/pots. If undamaged, displaced and/or returned to suitable sediment, the three main sea-pens found in this biotope were shown to recover to an upright position relatively quickly (24 to 72 hours) (Hill et al., 2020). *V. mirabilis* and *P. phosphorea* can avoid abrasion by withdrawing into the sediment but a frequent disturbance will probably reduce feeding time and viability. However, *F. quadrangularis* cannot withdraw, is the tallest of the three of sea-pens (up to 2 m) and is the most likely to be displaced or removed by surface abrasion and towed gear, so authors suggested a resistance of 'Low' where it is a dominant member of the community (Hill et al., 2020). Both *P. phosphorea* and *F. quadrangularis* were able to reinsert themselves into the sediment if removed as long as the peduncle remained in contact with the sediment surface, except in one specimen in which the peduncle was damaged (Kinnear et al., 1996).

4.3.2 Removal of target species

A major fishery exists for *N. Norvegicus*, which is a characterising species of the sea-pen and burrowing megafauna communities feature (Hill et al., 2020). *N. Norvegicus* is fished throughout most of the geographic range of the biotopes in which it occurs (Hill et al., 2020). Across the range, a small proportion (< 5 %) are fished using traps (Ungfors et al., 2013). The physical effects of traps on seabed communities are addressed in section 4.3.1. This pressure addresses the direct removal or harvesting of biota. Ecological consequences of this include the

sustainability of stocks, impacting energy flows through food webs and the size and age composition within fish stocks.

N. Norvegicus are opportunistic feeders that primarily consume crustaceans, molluscs and to a lesser extent polychaetes and echinoderms (Parslow-Williams et al., 2001). They are preyed upon by numerous bottom-feeding white fish including cod, haddock, skate and dogfish (Hill et al., 2020). They also support a variety of non-commercial species (including *Lesueurigobius friesii*, *Cycliophoran Symbion pandora*, *Balanus crenatus*, *Triticella koreni*, *Electra pilosa*, *Eudendrium capillare*, *Sabella pavonine* and *Serpula vermicularis*) by providing habitat either within their burrows or on themselves (Sabatini and Hill, 2008). Unsustainable removal of *N. Norvegicus* could therefore have an impact on the populations of these species.

Male *N. Norvegicus* are consistently landed by traps in larger numbers than females, although sex ratio does vary (Ungfors et al., 2013). This is likely because egg-bearing females are more prone to remain in their burrows for months at a time and in laboratory conditions, large males are less inclined to make burrows (Sabatini and Hill, 2008; Ungfors et al., 2013). Trap fisheries are characterised by larger sized *N. Norvegicus* that gain a higher price than in individuals landed by trawl fisheries (Ungfors et al., 2013). This is because animals need time to explore the trap and discover its entrances. Larger males spend longer out of their burrows than smaller individuals and so increase their opportunity to be attracted to the trap's bait and begin their exploration of the trap (Briggs, 1995). Smaller *N. Norvegicus* have also been observed being chased away from the trap by various competitors or more cautious of the gear itself (Bjordal, 1986). Trap fisheries for *N. Norvegicus* also have lower discard and by-catches of ground fish with lower mortalities than trawl fisheries (Ungfors et al., 2013). Unsustainable removal could impact the size and age composition of *N. Norvegicus* stocks.

JNCC and Natural England's AoO (JNCC, 2018a, 2018b, 2021b, 2021c; Natural England, 2022b) have classed the feature as not sensitive to the ecological effects of this pressure but have advised that it should still be considered at this stage because the pressure could pose a risk at a site level.

4.3.3 Removal of non-target species

Observations by Eno et al. (2001) have shown that traps can uproot *P. phosphorea*, possibly because it enters the trap through its netting following its initial retraction as the trap is set, causing it to be removed from the sediment as the trap is hauled (Adey, 2007). A study assessing bycatch in Alaska found that sea-pens were recorded in 0.3% of trap bycatch (Stone et al., 2015). Adey (2007) observed fishing trips on commercial *Nephrops* trap vessels on the West coast of Scotland and found that there is a moderate quantity of *V. mirabilis* and *P. phosphorea* present in the bycatch. Discard survival for these species was determined to be high because there are high concentrations of *V. mirabilis* and *P. phosphorea* in areas with high levels of

trap fishing activity. On the other hand, *F. quadrangulis* bycatch was very low in spite of high densities in areas intensely fished by the *Nephrops* trap fishery (Adey, 2007). Adey (2007) concludes that traps do not have an adverse effect on these three species.

Adey (2007) found that for every 1 kg of *N. Norvegicus* landed from creels, there was 1.1 kg of general bycatch and 0.2 kg *N. Norvegicus* discarded. This general bycatch included large biomasses of *P. bernhardus* and *L. depurator*, which are characterising species of the sea-pen and burrowing megafauna communities feature. Because organisms normally remain uninjured in the trap, it is likely that a high proportion of these crustacean species survived, once returned to the sea (Suuronen et al., 2012). Of the *N. Norvegicus* discarded, 8.6% were taken by marine birds, with this figure increasing significantly with wind strength (Adey, 2007). Trap bycatch tends to be discarded in the same area it was caught with minimal harm inflicted (Adey, 2007). A key consideration for *N. Norvegicus* bycatch survivability is the light-induced damage sustained to their eye when exposed to light for even short periods of time (<1 min), though previous studies have shown that blind *N. Norvegicus* survive, grow and reproduce successfully, with loss of sight having little effect on behaviour (Gaten et al., 1990; Chapman et al., 2000; Adey, 2007). Adey (2007) observed trap caught *N. Norvegicus* behaviour for 10 minutes after they were returned to the seabed. *N. Norvegicus* behaviour did not appear to be affected following capture and return to the seabed though few observations were made in this study, the author concludes that there is limited evidence pointing to high mortality of *N. Norvegicus* in the trap fishery and that most bycatch returns unharmed. The mortality of incidental catch in traps is generally considered low due to the selectivity for the target species and high probability of survival for any unwanted species caught and returned (Broadhurst et al., 2006; Suuronen et al., 2012).

4.4 Variation in impacts

Available literature suggests that traps likely do not pose a significant risk to sea-pen and burrowing megafauna communities. Potential impacts of traps may vary with fishing intensity (Kaiser, 2014), environmental factors such as the suitability of the substrate that an uprooted sea-pen is relocated to, the weather conditions the gear is subject to whilst in the water (for example, tides, currents and storm activity) (Sewell and Hiscock, 2005), the weather conditions when removing the traps, the ecology and life history stages of different species within the communities (Hughes, 1998). For example, life history characteristics of the crustacean species *C. macandreae* are that it is long-lived, reaches sexual maturity after five years, has a low fecundity and lacks a pelagic stage and therefore has low resilience to damage from fishing gear (Hill et al., 2020). Seasonality of breeding and larval recruitment causes variation in population sizes throughout the year in some species in the biotope, which may make a species more likely to be impacted by traps during particular

months (Hill et al., 2020). Additionally, impacts will vary depending on if species such as *Nephrops norvegicus* is caught by traps when individuals are bearing eggs, or before they have reached sexual maturity and had a chance to reproduce (Hill et al., 2020). Abilities of some species reduce the impacts of traps, for example *Virgularia mirabilis* can withdraw into its burrow rapidly so cannot be uprooted by dragged traps (Eno et al., 2001).

Additionally, although experimental studies have looked at the short-term impacts of traps on sea-pens, the potential long-term effect of repeated impacts remains unknown. It is possible that the continuous impact on sea-pens from traps could cause a gradual, cumulative deterioration in condition (Eno et al., 2001).

The feature sea-pen and burrowing megafauna communities can be comprised of the following biotopes: sea-pen and burrowing megafauna in circalittoral fine mud (and its sub-biotope: sea-pens, including *Funiculina quadrangularis*, and burrowing megafauna in undisturbed circalittoral fine mud); burrowing megafauna *Maxmuelleria lankesteri* in circalittoral mud; *Brissopsis lyrifera* and *Amphiura chiajei* in circalittoral mud; and *Atrina fragilis* and echinoderms on circalittoral mud. These biotopes all have slightly different characterising species and therefore different sensitivities to various pressures. These variations will be addressed at the site level assessment.

4.5 Summary of the effects of traps on sea-pen and burrowing megafauna communities

Literature suggests that traps are unlikely to significantly impact sea-pen and burrowing megafauna communities, however a site level assessment considering the site conservation objectives, intensity of fishing activity taking place and exposure to natural disturbance will be needed to determine whether management will be required.

The site level assessment will assess fishing activities for their impact upon protected habitats and species (in this case, the relevant biotopes for sea-pen and burrowing megafauna communities). Specifically, this assessment considers the potential for these activities to hinder the conservation objectives of the MCZ. The data used in the assessment will include vessel monitoring system (VMS) data, as well as feature habitat data from JNCC and Natural England. Where the assessment concludes that the current levels of management is not sufficient to protect the designated features of the site, recommended management options will be provided. MMO has regard to the best available evidence and through consultation with relevant advisors, stakeholders, and the public, will conclude which management option is implemented.

Management of the interaction between traps and sea-pen and burrowing megafauna communities may be unnecessary for MPAs designated for this feature. In which case, a site monitoring and control plan, including regular monitoring of this fishing activity with no restrictions, may be suggested to be sufficient at this stage.

5 Fan mussel

This section brings together and analyses the available evidence on how traps affect fan mussel communities.

Fan mussel (*Atrina fragilis*, family: *Pinnidae*) is a designated feature of the following MCZs: East of Haig Fras (JNCC, 2021d), South of Isles of Scilly (JNCC, 2021h) and South West Deep (West) (JNCC, 2018j).

Fan mussel is distributed throughout UK continental shelf waters (Tyler-Walters and Wilding, 2022), particularly in deep waters around the Shetland Isles and Orkney, the West Coast of Scotland, possibly the North-East of Scotland, the South Coast of England (particularly around Cornwall), the Channel Isles, Pembrokeshire and Northern Ireland (Solandt, 2003; Tyler-Walters and Wilding, 2022).

In the UK, fan mussel is often found as solitary individuals, but can also occur as small groups or patches of individuals forming small beds (Tyler-Walters and Wilding, 2022). This species is generally found in mud, sandy mud and fine gravel habitats, particularly in full salinity sheltered areas with weak to moderately strong tidal flows (Tyler-Walters and Wilding, 2022). Their distribution has been linked to several environmental variables including depth, seabed topography, current speed, and percentage of mud and gravel (Stirling, 2016).

Information on life history, biology and reproductivity of fan mussel is sparse (Stirling, 2016); however, some details can be inferred with caution from the wider *Pinnidae* family (Solandt, 2003). Due to its large size and based on growth rates of 3 to 4 cm per year, fan mussel is thought to be a long-lived species with larger specimens being at least 10 to 12 years old (Solandt, 2003), with some sources estimating lifespans of 20 to 100 years old (Tyler-Walters and Wilding, 2022). Based on information from similar species, fan mussel is cautiously presumed to be gonochoristic (have two sexes), be broadcast spawners (release female and male gametes into the water column where fertilisation occurs) and have free-swimming, plankton-feeding larvae (Stirling, 2016).

However, insufficient information is available to estimate age of maturity, reproductive frequency, fecundity, the reproductive season and location, generation time, duration of larval stage or larval settlement period (Tyler-Walters and Wilding, 2022). Spawning for fan mussel appears similar to other pinnids in temperate latitudes, peaking over summer and winter (Stirling, 2016). Dispersion may occur over large distances due to their pelagic spawning life history (Tyler-Walters and Wilding, 2022).

5.1 Overview of the sensitivity of fan mussel to traps

5.1.1 Sensitivity – resistance to damage

Fan mussel has thin and brittle shells (Tyler-Walters and Wilding, 2022), making them very fragile and sensitive to physical and mechanical damage. Fishing gears can consequently damage the portions of the shell that protrude into the water column and, if the fishing gears (such as scallop dredges) penetrate the seabed, such gears can also damage the portions of shell embedded in the sediment (Fryganiotis et al., 2013; Stirling, 2016). Fan mussel may be able to adapt to such damage by withdrawing into the remaining undamaged shell whilst the damaged shell is repaired at a rate of approximately 1 cm per year (Solandt, 2003). Post-larval pinnids have small shells (1 to 2 cm) that are easily damaged and weakly attached to the substrate (Stirling, 2016). Being partly buried in the sediment, fan mussel is also sensitive to being dislodged and removed from the substrate (Stirling, 2016). Individuals are unable to re-burrow themselves following a disturbance incident (Hiscock and Jones, 2004). Despite being able to burrow vertically they cannot right themselves if removed from the sediment and laid on their sides (Yonge, 1953 cited in Tyler-Walters and Wilding, 2017). Whole populations may be removed if sediment is removed to a depth of 30 cm (Tyler-Walters and Wilding, 2022).

5.1.2 Recovery – rate of recovery

Fan mussel recoverability may be limited by their life history characteristics (Tyler-Walters and Wilding, 2017). Long lifespans, slow growth, low gamete production and sporadic recruitment reduces their ability to recover from damage, displacement, or mortality (Hiscock and Jones, 2004; UK Biodiversity Group, 1999). There is however still a major lack of information on fan mussel life history which adds to the degree of caution that needs to be taken when assessing the recoverability of the species as a whole.

Larval dispersal may be limited or irregular (Tyler-Walters and Wilding, 2022) and larvae mortality is likely to be high (Stirling, 2016), possibly due to an infrequency of suitable conditions (UK Biodiversity Group, 1999). Fan mussel recruitment is likely poorer and more variable than other bivalve species (UK Biodiversity Group, 1999), however recruitment levels may be higher at locations with inlets and embayments where larvae are entrapped. With patchy, low-density populations, fertilisation is also likely to be inefficient (Tyler-Walters and Wilding, 2022).

Pinnids have fast shell growth rates relative to other bivalves (Stirling, 2016); however, growth rates are likely slower for sexually mature individuals, which must put energetic resources into gonad development rather than shell accretion. Shell growth rates will also vary with location, water temperature, and availability of food supply (Solandt, 2003; Tyler-Walters and Wilding, 2022). An under-recording of the species in deep waters suggests that the species may be more prevalent in deeper waters than previously realised, and thus deep-waters may provide a potential

reservoir for recruitment; however, there is no evidence to support this (Tyler-Walters and Wilding, 2022).

Slow recovery rates may be a contributing factor to the decline of fan mussel in UK inshore waters over the last hundred years (Solandt, 2003; Tyler-Walters and Wilding, 2022). In summary, the recruitment and recovery of fan mussel is likely to be prolonged and may take up to 25 years in the UK where populations are sparsely distributed (Tyler-Walters and Wilding, 2022). The species is categorized as having low resilience to any loss of population or 'very low' resilience to severe declines in population abundance (Tyler-Walters and Wilding, 2022).

5.2 Level of literature, caveats and assumptions

Biology and distribution data for fan mussel is generally limited (Fryganiotis et al., 2013; Stirling, 2016), however information about suitable habitats is available so assumptions can be made about potential impacts to this species in certain areas. There is limited evidence regarding fishing impacts specific to fan mussel and therefore evidence from other species within the Pinnidae family has been cautiously considered in some cases. It should however be noted that there is no true proxy species for fan mussel and that species considered in the Pinnidae family occur in a different climate to England.

There is limited information on the impacts from traps and therefore evidence regarding anchor impacts and trampling has been considered, with assumptions made that these impacts may be similar. There is no record or evidence of fan mussel being removed or damaged by traps.

5.3 The pressures of traps on fan mussel

As a result of using traps, this feature may be sensitive to the following pressures, so they are considered in this document:

- abrasion or disturbance of the substrate on the surface of the seabed
- penetration and/or disturbance of the substrate below the surface of the seabed, including abrasion
- removal of non-target species

There is insufficient evidence available to determine whether this feature is sensitive to the following pressures as a result of the use of traps:

- hydrocarbon and PAH contamination
- introduction or spread of invasive non-indigenous species
- litter
- organic enrichment
- synthetic compound contamination
- transition elements and organo-metal contamination

5.3.1 Abrasion or disturbance of the substrate on the surface of the seabed and penetration and/or disturbance of the substratum below the surface of the seabed, including abrasion

These pressures are grouped together to avoid repetition, due to the similar nature of their impacts on the species. Fan mussel typically live in the sublittoral fringe, in subtidal mud, sandy mud or gravel habitats (Tyler-Walters and Wilding, 2022). Traps are not generally considered a fishing activity that penetrates the seabed (Hall et al., 2008), and abrasion and penetration towards sediment habitats will be more significant in bottom towed gears. However, impacts from traps are still possible through the interaction between the seabed and the gear itself, including associated lines and anchors.

This is more likely to occur if the gear moves across the seabed during hauling of gear or when the gear is subject to strong tides, currents, or storm activity. There is limited direct evidence of the impacts of traps on subtidal sediments, however their impacts on the physical structure of subtidal mud, sandy mud or gravel habitats are discussed in the sediments and sandbanks review in section 9.

5.3.2 Removal of non-target species

Fragile infaunal species which live on or within the surface sediments (such as bivalves, holothurians, gastropods) are particularly sensitive to damage or disturbance (Kaiser and Spencer, 1996). If removed from the sediment, fan mussels are unable to re-burrow into the substrate. Despite being able to burrow vertically they cannot right themselves if removed from the sediment and laid on their sides (Yonge, 1953 cited in Tyler-Walters and Wilding, 2017). If traps are set directly onto this species, there is the potential for individuals to be dislodged, however the removal of fan mussel from the sediment by traps is very unlikely.

Fan mussel has a fragile shell which is thought to be easily damaged by anchor impacts or trampling (Tyler-Walters and Wilding, 2022). Conversely, the mantle and ctenidia can be withdrawn into the shell and a damaged edge of the shell repairs quickly (Solandt, 2003; Yonge, 1953 cited in (Tyler-Walters and Wilding, 2017).

This could potentially reduce its sensitivity to low levels of abrasion, for example from trampling or possibly traps that damage exposed shell (Tyler-Walters and Wilding, 2022), however the biological cost of repairs is likely to be high. Traps are not known to target fan mussel in UK waters and there is also no evidence or records of individuals being caught as bycatch by traps. This evidence suggests that traps do not pose a significant risk towards fan mussel through removal of non-target species.

5.4 Variation in impacts

Although traps likely do not pose a significant risk to fan mussel, any potential impacts of traps may vary with fishing activity, environmental factors and the ecology and life history stage of this species. Fishing intensity in particular may drive

potential impacts. Based on a matrix approach using both scientific literature and expert judgement, sedimentary habitats containing long-lived bivalves are classed as having medium sensitivity to high intensity levels of potting (Hall et al., 2008; Eno et al., 2013).

The distribution of fan mussel is linked to several environmental parameters, which may in-turn influence spatial overlap with potting fisheries (Stirling, 2016). Growth rates (and thus potentially recovery from abrasion impacts) could vary with life history stage, location, temperature, and food supply (Solandt, 2003; Tyler-Walters and Wilding, 2022). Levels of natural disturbance might influence potential impacts, with high natural disturbance potentially increasing the likelihood of gear components being moved across the seabed and thus potentially increasing the probability of snagging protruding fan mussel shells.

As a sessile benthic species (Stirling, 2016), the spatial overlap between potting activity and the distribution and abundance of fan mussel populations will clearly influence pathways for impact. Recoverability from disturbance will also be influenced by population density, with sparser populations having lower fertilisation efficiency (Tyler-Walters and Wilding, 2022).

5.5 Summary of the effects of traps on fan mussel

The literature suggests that traps are unlikely to have a significant impact on fan mussel, however a site level assessment considering the fishing activities taking place and site conservation objectives will be needed to determine whether management will be required.

The site level assessment will assess fishing activities for their impact upon protected habitats and species. Specifically, this assessment considers the potential for these activities to hinder the conservation objectives of the MCZ. The data used in the assessment will include VMS data, as well as feature habitat data from JNCC and Natural England. Where the assessment concludes that the current level of management is not sufficient to protect the designated features of the site, recommended management options will be provided. MMO has regard to the best available evidence and through consultation with relevant advisors, stakeholders, and the public, will conclude which management option is implemented.

Using scientific literature and expert judgement, sedimentary habitats containing long-lived bivalves have been categorised in the available literature as having a medium sensitivity to high intensity potting (classed as intensities seen in the heaviest of fisheries, for example 5 pots lifted daily per hectare, 100 m by 100 m) and otherwise have low sensitivity to potting (Hall et al., 2008; Eno et al., 2013). As fan mussel is a long-lived species found primarily in sedimentary habitats, such thresholds could be used to inform site level assessments and to determine whether management will be required.

6 Ocean quahog

This section brings together and analyses the available evidence on how traps affect ocean quahog.

Ocean quahog (*Arctica islandica*) is a long-lived bivalve mollusc found throughout the continental shelf area of English waters, as well as offshore. Ocean quahog is a designated feature of the following MCZs: North East of Farnes Deep (JNCC, 2018h), Fulmar (JNCC, 2021e), Holderness Offshore (JNCC, 2021f) and Farnes East (JNCC, 2017a).

Ocean quahog is designated as a species of conservation importance in English and Welsh waters and has been recorded from the Baltic, Iceland, the Faroe Islands and throughout the continental shelf of the North Atlantic (Witbaard and Bergman, 2003). The depths at which it can be found range from the low intertidal zone at 4 to 480 m, but most commonly between 10 to 280 m (Holmes et al., 2003). Ocean quahog is known to occur in waters with salinity of 16 to 40 practical salinity units (PSU) and temperatures of 6 °C to 16 °C, although experiments have recorded tolerance of up to 20 °C for a limited period of time (Oeschger and Storey, 1993; OSPAR, 2009; Tyler-Walters and Sabatini, 2017). The last remaining extant species of the family Arctidae (Morton, 2011), ocean quahog is considered the longest living non-colonial animal and is capable of living for centuries.

6.1 Overview of the sensitivity of ocean quahog to traps

6.1.1 Sensitivity – resistance to damage

A long generation time of approximately 83 years (Hennen, 2015), low growth rate in adults, variable age and size at maturity, and unpredictable recruitment success (owing to variable environmental factors, a long planktonic larval stage and low rates of juvenile survival), mean that ocean quahog is particularly sensitive to pressures exerted by fishing activity (OSPAR Commission, 2009). Additionally, population structure can be skewed, with some areas being dominated by adults and others by juveniles (AquaSense, 2001).

MarLIN has assessed the species as having varying resilience depending on location and amount of mortality. If a population has experienced significant mortality, then a precautionary resistance of 'Very Low' is recorded, as recovery is likely to take more than ten years, or potentially in excess of 25 years (for example in the North Sea; Witbaard and Bergman, 2003). If a population has only suffered some mortality, then the species is assessed as having a resilience of 'Medium' as recovery may be possible from low levels of continuous recruitment (Tyler-Walters and Sabatini, 2017). For definitions of resistance (tolerance), resilience (recovery) and sensitivity rankings from the MarESA (Tyler-Walters et al., 2018), see the glossary in the [Stage 3 Call for Evidence Introduction](#).

There is a lack of literature describing the sensitivity of the species to impacts associated with the use of traps, however there is evidence of the impacts from bottom towed gear use. There is significant evidence of the impacts of bottom trawling on ocean quahog in the North Sea, with benthic surveys indicating a reduction in distribution of the species between 1902 and 1986 and a reduction in species abundance between 1972 and 1980 and then between 1990 and 1994 (Rumohr et al., 1998). Gilkinson et al. (1998) noted that a key factor in determining sensitivity of bivalves to bottom trawling activity is burial depth, combined with size. Bivalves close to the sediment surface that are buried deep enough to establish stability within the sediment are reported to be more likely to break when they come into contact with otter trawls as they are less likely to be excavated to the surface without damage. However, bivalves that are excavated to the surface by bottom towed gear activity become increasingly exposed to indirect mortality via predation (Ragnarsson et al., 2015).

There is a lack of literature describing the impacts of traps and on ocean quahog. Although these gear types can cause some abrasion of the seabed (Roberts et al., 2010), given the hard shell of ocean quahog and limited seabed contact of these gears, they are unlikely to significantly impact the species.

The recruitment of ocean quahog is linked to water temperature, with increasing temperatures being attributed to the cause of low recruitment success in North Sea populations (Witbaard and Bergman, 2003). With increasing warming of oceans, southerly populations of ocean quahog may experience recruitment failure which could result in range contraction of the species and therefore a change in the sensitivity of the species to fishing activity.

6.1.2 Recovery – rate of recovery

Recovery from damaging activities will depend on the intensity and frequency of the impact and the recruitment processes of a species. There is limited research that has examined the recovery of ocean quahog; however, it is thought that their recovery may be limited by their life history characteristics of having long lifespans, slow growth rates and taking 5 to 15 years to reach maturity (Tyler-Walters and Sabatini, 2017).

It has been reported that reductions in adult ocean quahog density over fished grounds can negatively affect recovery via less effective recruitment (Witbaard and Bergman, 2003). The minimum required density of ocean quahog for reproductive success is not currently known (Hennen, 2015) therefore precautionary management approaches may be required in order to ensure that ocean quahog density does not fall below the level required to sustain the population via sexual reproduction. As ocean quahog populations are potentially reproductively isolated from each other (Holmes et al., 2003), recovery may vary at a population level. A low and constant rate of recruitment may be sufficient for ocean quahog populations to recover from

low to moderate disturbance; however, it may be difficult for ocean quahog to recover from a sustained high level of fishing (Tyler-Walters and Sabatini, 2017).

It has been suggested that UK waters may be a sink of new ocean quahog recruits from Iceland, with long periods without successful recruitment in between larval settlement events (Witbaard and Bergman, 2003). Larvae are thought to be brought down the east coast of the UK and into the mid and southern North Sea by slower moving waters inside gyres that allow settlement to happen. The recovery of ocean quahog populations at a site is likely to depend on an outside source of larvae that arrives infrequently and unpredictably. The recovery of the species is also highly dependent on larger scale environmental pressures such as climate change (JNCC, 2018m).

6.2 Level of literature, caveats and assumptions

There is no evidence on the impacts of traps on ocean quahog. Due to the unique life history traits and characteristics of ocean quahog, there are no proxy species with which to assess the impacts of traps. As a result, there is a considerable amount of uncertainty with regards to the impacts of traps on ocean quahog.

6.3 The pressures of traps on ocean quahog

As a result of traps, this feature may be sensitive to the following pressures, so they are considered in this document:

- abrasion or disturbance of the substrate on the surface of the seabed

There is insufficient evidence available to determine whether this feature is sensitive to the following pressures as a result of the use of traps:

- hydrocarbon and PAH contamination
- introduction or spread of invasive non-indigenous species
- litter
- synthetic compound contamination
- transition elements and organo-metal contamination.

6.3.1 Abrasion or disturbance of the substrate on the surface of the seabed

Ocean quahog typically live in sublittoral firm sediments including level offshore areas, buried or partially buried in sand and muddy sand (Tyler-Walters and Sabatini, 2017). Traps are not generally considered a fishing activity that penetrates the seabed (Hall et al., 2008). However impacts from traps are still possible through the interaction between the seabed and the gear itself including associated lines and anchors. Abrasion is most likely to occur when the gear moves across the seabed during hauling of gear or when the gear is subject to strong tides, currents, or storm activity. There is no direct evidence of the impacts of traps on subtidal sediments,

however they are unlikely to significantly impact the physical structure of sand or muddy sand.

Benthic species which live on or within the surface sediments (such as bivalves, holothurians, gastropods) are particularly vulnerable to damage or disturbance (Kaiser and Spencer, 1996). The depth that ocean quahog can be found at ranges from the low intertidal zone at 4 m to 480 m (Holmes et al., 2003). Ocean quahog live buried vertically in the top layers of sediment (soft sands and muddy sands). Ocean quahog may be just below the sediment surface or buried in up to 14 cm of sediment, with its siphons protruding from the sediment surface (Strahl et al., 2011). It is therefore possible for ocean quahog siphons to be damaged by traps.

Additionally, traps are not known to target ocean quahog in UK waters and there is no evidence of individuals being caught as bycatch by traps.

6.4 Variation in impacts

Although traps likely do not pose a significant risk to ocean quahog, any potential impacts of traps may vary with fishing activity, environmental factors and the ecology and life history stage of this species. Fishing intensity in particular may drive potential impacts, with sedimentary habitats containing long-lived bivalves having medium sensitivity to high intensity levels of potting (Hall et al., 2008; Eno et al., 2013).

Growth rates of ocean quahog (and thus potentially recovery from abrasion impacts) also vary with location, temperature, and food supply (Tyler-Walters and Sabatini, 2017). It should be noted that ocean quahog is a very slow growing organism even when growth rates are at the higher end of the spectrum (average 1.5 mm per year; Cargnelli et al. (1999)). The age dynamics of a population of ocean quahog may affect their sensitivity to traps, as shell strength and burial depth in the sediment varies with age. Some studies suggest larger, older individuals to be more susceptible to damage due to a comparatively lower ratio of shell thickness to shell size than juveniles (Rumohr and Krost, 1991). Whereas other studies suggest the shells of older individuals to typically be thicker and therefore provide a higher level of protection (Hawkins and Angus, 1986).

Levels of natural disturbance might influence potential impacts, with high natural disturbance potentially increasing the likelihood of gear components being moved across the seabed and thus potentially increasing the probability of damaging ocean quahog shells. The spatial overlap between potting activity and the distribution and abundance of ocean quahog populations will influence the pathways for impact.

6.5 Summary of the effects of traps on ocean quahog

Traps are unlikely to have a significant impact on ocean quahog, however a site level assessment considering the site conservation objectives, intensity of fishing activity

taking place and exposure to natural disturbance will be needed to determine whether management will be required.

The site level assessment will assess fishing activities for their impact upon protected habitats and species. Specifically, this assessment considers the potential for these activities to hinder the conservation objectives of the MCZ. The data used in the assessment will include VMS data, as well as feature habitat data from JNCC and Natural England. Where the assessment concludes that the current level of management is not sufficient to protect the designated features of the site, recommended management options will be provided. MMO has regard to the best available evidence and through consultation with relevant advisors, stakeholders, and the public, will conclude which management option is implemented.

Using scientific literature and expert judgement, sedimentary habitats containing long-lived bivalves have medium sensitivity to high intensity potting (classed as intensities seen in the heaviest of fisheries, for example 5 pots lifted daily per hectare, 100 m by 100 m) and otherwise have low sensitivity to potting (Hall et al., 2008; Eno et al., 2013). As ocean quahog is a long-lived species found primarily in sedimentary habitats, such thresholds could be used to inform site level assessments and to determine whether management will be required.

Management of the interaction between traps and ocean quahog may be unnecessary for MPAs designated for this feature.

7 Rocky reef

This section brings together and analyses the available evidence on how traps affect rocky reef features.

Reefs are an Annex I habitat listed in the Council Directive 92/43/EEC (the Habitats Directive). Several MCZ features including circalittoral and infralittoral rock, subtidal chalk, and Ross and honeycomb worm reefs correspond to the Annex I reef classification. JNCC classifies reef into one (or more) of the following three subtypes: bedrock, stony and biogenic (JNCC, 2019). For the purpose of this review, MMO has separated reefs into two different categories; bedrock reef and stony reef are categorised as 'rocky reef' and biogenic reef is categorised as 'biogenic reef (*Sabellaria spp.*)'. This section only refers to rocky reef.

For special areas of conservation (SACs), bedrock and stony reef are the terms used for designated features. For MCZs, the equivalent is circalittoral and infralittoral rock (high, moderate, and low energy). Low energy circalittoral rock and subtidal chalk reef have not been included in this review as they are not designated features of any of the relevant sites. Intertidal and infralittoral rock have also not been included in this review because they are not located within the relevant sections of the MPAs, where MMO is the principal regulator for fishing.

Rocky reef features as considered in this review, are found in the following MCZs: Farnes East (JNCC, 2017a), Goodwin Sands (Natural England, 2021), Cape Bank (DEFRA et al., 2019), Hartland Point to Tintagel (Natural England, 2022d), South of Celtic Deep (JNCC, 2021g), Foreland (Natural England, 2022c), East of Haig Fras (JNCC, 2021d), Offshore Brighton (JNCC, 2018i); and the following SACs: Start Point to Plymouth Sound and Eddystone (Natural England, 2018b), Land's End and Cape Bank (Natural England, 2018a), Haig Fras (JNCC, 2018d), Wight-Barfleur Reef (JNCC, 2018l).

Pink sea fan and fragile sponge and anthozoan communities are present in one Stage 3 site (Hartland Point to Tintagel) and will be considered in this review. Pink sea fan and fragile sponge and anthozoan communities are similar to some of the biotopes associated with rocky reefs and therefore fisheries impacts will likely be similar. They are also often found overlaid on rocky reefs creating a mosaic of multiple features. For management purposes, these mosaic habitats are therefore considered as one feature.

Rocky reef is recognised as areas where animal and plant communities develop on rock (bedrock) or stable boulders and cobbles (stony). Rocky reefs are defined by Irving (Irving, 2009) as 'hard compact substrata on solid and soft bottoms, which arise from the sea floor in the sublittoral and littoral zone. Reefs may support a zonation of benthic communities of algal and animal species.'

The sublittoral zone (extending from the lowest limit of the intertidal to the outer edge of the continental slope) can be divided into the infralittoral zone (characterised by algae) and the circalittoral zone (the subzone below the infralittoral dominated by animals) (JNCC, 2022). Both bedrock reef and stony reef are assigned one of three energy levels (i.e., high, moderate, or low energy, depending on exposure to tidal and wave energy) and are associated with rocky reefs (Natural England, 2015). Rocky reef sub-features found in Stage 3 sites include high and moderate energy circalittoral rock.

Rocky reefs can be present in a wide range of topographical forms, ranging from vertical rock walls to horizontal ledges, sloping or flat bedrock, broken rock, boulder fields, and aggregations of cobbles (JNCC, 2021a). These reefs are characterised by communities of attached algae and invertebrates, usually associated with a range of mobile animals, including invertebrates and fish (JNCC, 2021a). Rocky reefs provide structural complexity for many sensitive and diverse epifauna and such habitats may be vulnerable to sporadic or prolonged pressures from fishing activities and associated gears (Sangil et al., 2013; Kaiser, 2014; Gall et al., 2020).

Consequentially, the short and long-term effects are wide ranging; such impacts may reduce species composition, biomass, and diversity, potentially resulting in removal of key species and thereby leading to changes in ecosystem functionality and resilience over different timescales (Gall et al., 2020).

7.1 Overview of the sensitivity of rocky reef to traps

7.1.1 Sensitivity – resistance to damage

Numerous different biotopes can make up the high and moderate energy circalittoral rock habitats (EEA, 2012) making the sensitivity of rocky reef habitats highly variable. The sensitivity of each biotope to different pressures has been assessed following the MarESA approach (Tyler-Walters et al., 2018). Individual biotope sensitivities range from low to high. This range in sensitivity is caused by the range in species that make up each biotope and the different hydrological conditions in which they occur. These biotope sensitivities are then used in JNCC and Natural England's advice on operations (AoO) to determine the site level sensitivity of the designated habitat feature (in this case, high or moderate energy circalittoral rock) to various activities (in this case, traps). Biotopes with the highest sensitivity to the relevant physical pressure caused by these fishing gears (abrasion/disturbance of the surface of the substratum or seabed) are:

- deep sponge communities (Readman, 2018a) - whilst some of the characterising sponges can be quite elastic, abrasion pressures, especially by heavy gears, have been shown to cause significant damage to the sessile epifaunal sponges. Therefore, sensitivity is assessed as 'High'
- chalice sponge (*Phakellia ventilabrum*) and axinellid sponges on deep, wave-exposed circalittoral rock (Readman, 2018b) - as abrasion pressures, especially by heavy gears, have been shown to cause significant damage to the sessile epifaunal sponges, sensitivity is assessed as 'High'
- pink sea fan (*Eunicella verrucosa*) and Ross coral (*Pentapora foliacea*) on wave-exposed circalittoral rock (Readman et al., 2018) - *E. verrucosa* is a sessile epifauna and is likely to be severely damaged by heavy gears, such as scallop dredging (MacDonald et al., 1996). However, some studies suggest that the species may be more resistant, particularly to low intensity lighter abrasion pressures, such as traps and associated anchor damage (Eno et al., 1996). Taking all the evidence into account, sensitivity is assessed as 'High'
- circalittoral caves and overhangs (Readman and Hiscock, 2018) - as abrasion pressures, especially by heavy gears, have been shown to cause significant damage to the sessile epifaunal sponges. Although the biotope's occurrence on cave walls and ceiling, and overhangs may protect the habitat from trawling, it may be impacted by mooring chains or abraded by anthropogenic debris. Therefore, a precautionary sensitivity of 'High' is suggested
- sponges, cup corals and anthozoans on shaded or overhanging circalittoral rock (Readman, 2018c) - as abrasion pressures, especially by heavy gears, have been shown to cause significant damage to the sessile epifaunal sponges. Although the biotope's occurrence on cave walls and ceiling, and overhangs may protect the habitat from trawling, it may be impacted by

mooring chains or abraded by anthropogenic debris. Therefore, a precautionary sensitivity of 'High' is suggested.

Sensitivity assessments suggest there is the potential for static gear such as traps to cause damage to rocky reefs and sensitive epifauna (Eno et al., 2013). Vertical rock with associated species were shown to be highly sensitive to traps only at moderate-heavy fishing intensity (Eno et al., 2013). Rock with low-lying fast-growing faunal turf were shown to have medium sensitivity to traps at high fishing intensity (Eno et al., 2013). Rock with erect and branching species were shown to have medium sensitivity to traps at moderate-heavy fishing intensity (Eno et al., 2013). These assessments allocated resistance and resilience scores to derive sensitivity by using the best available information that may or may not have been supported by empirical evidence from well-designed experimental studies (Eno et al., 2013). Empirical studies on the other hand have had mixed results with some finding evidence that rocky reef habitats and their communities are relatively unaffected by potting (Eno et al., 2001; Coleman et al., 2013; Haynes, et al., 2014). JNCC and Natural England (JNCC and Natural England, 2011) advised that the impacts of weights and anchors associated with static gear and hauling of gear can damage some species within fragile sponge and anthozoan communities on subtidal rocky habitats, but that other species appear to be resilient to individual fishing operations. They concluded that the sensitivity of these species to low intensity potting is low (JNCC and Natural England, 2011).

Rocky reef also has varying sensitivities to the biological effects of traps, such as removal of target and non-target species. Removal of characterising species may result in the loss of the biotope (Readman, 2018a) and the removal of commercial fishery species such as crustacea may impact the productivity and community composition of the reef feature (Babcock et al., 1999). These sensitivities will vary by biotope and fishing intensity.

7.1.2 Recovery – rate of recovery

Recovery rates for the habitats associated with sublittoral rock will depend on the species present. Recovery rates may vary with life-history characteristics, larval longevity, dispersal potential, recruitment, and growth rates (Kaiser et al., 2018). There is a lack of literature describing the recovery of the habitat from the use of traps, however there is available evidence of the recovery after the use of bottom towed gear. A study investigating the recovery of sessile epifauna following the exclusion of bottom towed gears in Lyme Bay, found that pink sea fan and Ross corals had projected recovery times of 17 to 20 years (Kaiser et al., 2018). Shorter-lived species such as dead man's fingers had much shorter recovery times of 2.5 to 6 years (Kaiser et al., 2018). The longevity of species will also influence recovery rates, with short-lived fauna (for example with lifespans of 1 to 3 years) potentially recovering from trawling in 0.5 to 3 years, whereas long-lived fauna (for example with lifespans > 10 years) may take several years (> 8 years) to recover (Hiddink et

al., 2019). The MarESA approach determined that the biotopes with the lowest resilience (recoverability) to the relevant physical pressure caused by these fishing gears (abrasion/disturbance of the surface of the substratum or seabed) are:

- deep sponge communities (Readman, 2018a) - as above in section 7.1.1. Therefore, resistance is assessed as 'Low' and resilience is assessed as 'Very Low'
- chalice sponge (*P. ventilabrum*) and axinellid sponges on deep, wave-exposed circalittoral rock (Readman, 2018b) - as above in section 7.1.1. Therefore, resistance is assessed as 'Low' and resilience is assessed as 'Very Low'
- pink sea fan (*E. verrucosa*) and Ross coral (*P. foliacea*) on wave-exposed circalittoral rock (Readman et al., 2018) - as above in Section 7.1.1. Therefore, resistance is assessed as 'Low' and resilience is 'Very Low'

7.2 Level of literature, caveats and assumptions

This review is based on information sourced from peer-reviewed scientific journals and research reports. The majority of literature covered in this section comes from the UK and the North Sea, whilst some of the research considered here derives from global reviews of the relevant literature (Jennings and Kaiser, 1998; Broadhurst et al., 2006; Suuronen et al., 2012). Some of the research related to the impacts of traps on soft coral and the movement of traps in storms comes from the tropical United States 'US' (Lewis et al., 2009; Shester and Micheli, 2011). While some metrics are presented to enumerate impacts, they do not necessarily reflect the impacts across the whole fishery.

Walmsley et al. (2015) undertook an extensive literature review on behalf of the Defra Marine Biodiversity Impact Evidence Group on the physical impacts of potting on a range of UK MPA features. Many of these studies are included in this literature review. Walmsley et al. (2015) noted a divergence between sensitivity assessments based on expert opinion that suggested certain reef species were sensitive to traps, and assessments based on empirical evidence that found very limited proof of negative impacts as a result of traps. Walmsley et al. (2015), however, did not take into account studies later than 2015, such as Rees (2018) which found that increased trap activity negatively impacted the recovery of sensitive reef species. These findings are discussed in detail in section 7.3.2. Walmsley et al. (2015) highlight there are limitations to all of the empirical studies reviewed, as well as gaps in evidence. The studies lacked longer-term research, unfished controls with similar physical and environmental conditions as the fished sites, or suitable environmental and pressure data to correlate with observed ecological data (Walmsley et al., 2015). Nevertheless, Walmsley et al. (2015) recommends that traps can be generally scoped out of assessments for bedrock and stony reef, unless there are site specific concerns around areas highlighted by some inconclusive empirical results or

sensitivity assessments or if trap intensity is deemed incompatible with the conservation objectives of the site.

7.3 The pressures of traps on rocky reef

As a result of traps, this feature may be sensitive to the following pressures, so they are considered in this document:

- abrasion or disturbance of the substrate on the surface of the seabed
- removal of target species
- removal of non-target species

There is insufficient evidence available to determine whether this feature is sensitive to the following pressures as a result of the use of traps:

- hydrocarbon and PAH contamination
- introduction of light
- litter
- synthetic compound contamination
- transition elements and organo-metal contamination
- underwater noise changes
- visual disturbance

7.3.1 Abrasion or disturbance of the substrate on the surface of the seabed

Abrasion impacts from traps may occur during deployment, positioning (via dragging), tidal/current movement and swell, and recovery (via hauling). Direct abrasive contact may occur from the trap itself; the end weight and anchors and indirect impacts may occur from scour, or the rubbing effects caused by the associated trap ropes (Hartnoll, 1998; Eno et al., 2001; Hall et al., 2008; Rees, 2018).

Commercial experiments conducted by Gall et al. (2020) found that rope movement once traps were deployed and left to soak in situ was observed for 51% of gear deployments, though in many of these instances (46 %) this movement was described as minimal, with no scour. Gall et al. (2020) also found that total possible seabed contact area (total distance travelled x area of base of trap) was $6.20 \text{ m}^2 \pm 0.61$, and the length of the seabed contact area (distance where trap contacted seabed x area of base of trap) was $3.04 \text{ m}^2 \pm 0.24$ (49.07 % of the total possible contact area). These results will change depending on the type of trap used, the site, and the target fishery, hence the need for site level assessments. This is discussed further in Section 7.4. In this study, one string of inkwell traps and one string of parlour traps were deployed over rocky reef habitats with four traps per string (Gall et al., 2020). The abrasion pressure is unlikely to impact the rocky substrate itself, being more likely to impact the taxa associated with the rocky reef habitats.

Mechanical impacts of static gear on rocky reef (for example weights and anchors hitting the seabed, hauling gear over seabed, rubbing/entangling effects of ropes) can damage some species (Eno et al., 1996). Other species have shown resilience to traps being dropped and hauled (Eno et al., 2001; Coleman et al., 2013) but there may still be an increased risk of cumulative damage from fishing activity at higher intensities (Eno et al., 2001).

Surveys of Land's End and Cape Bank were conducted in 2007 by Birchenough et al. (2008) and in August and October of 2010 by Axelsson and Dewey (2011). The results of these surveys were reviewed in a study of the MPA by Natural England (2012). Fishing activity in the inshore portion of the MPA is generally small scale and seasonal, consisting of traps, gill nets and handlines, and the exposure to physical damage through abrasion is therefore low (JNCC and Natural England, 2012). Exposure to abrasion in the offshore portion of the site is considered moderate due to potential trawling and dredge activity. There was a low level of trawling and dredging activity in the inshore section of the site due to a byelaw prohibiting the activity. The 2007 surveys recorded 'no evidence of habitat damage as a result of anthropogenic activity' (Axelsson and Dewey, 2011), the majority of which is static gear. The 2010 surveys recorded vertical rock faces occurring in several locations throughout the site colonised by large numbers of epifauna and Ross corals (Axelsson and Dewey, 2011). They found no evidence of damage caused by human activity (Axelsson and Dewey, 2011). When the video lines and photographs taken during the 2010 survey were compared alongside those of the 2007 surveys, Axelsson and Dewey (2011) found no evidence of changes to the biotope composition within the MPA. While the Axelsson and Dewey (2011) surveys cannot definitively rule out negative impacts from fishing activities, the absence of any evidence of physical damage, despite coinciding with higher levels of fishing effort than seen more recently, strongly indicates that fishing activities have resulted in minimal, if any, damage to the reef feature, including those biotopes and species identified as the most potentially sensitive. This is consistent with Coleman et al. (2013) who reported that there were no differences in ecological change from 2004 to 2007 between areas subject to controlled traps in a newly introduced no-take-zone, areas outside of the no-take-zone and to areas not fished at all.

Eno et al. (2001) observed the bending and recovery of pink sea fan following trap deployment and removal, which suggests minimal impact of traps on the species, however there may be more long-term implications to their ability to withstand immediate trap pressure. The pink sea fan skeleton is covered in soft tissue (coenenchyma) which may be damaged through these trap interactions. Fouling organisms such as the invasive warm water barnacle (*Hesperibalanus fallax* formerly *Solidobalanus fallax*) have been observed colonising pink sea fan on the Eddystone reef near Plymouth (Southward et al., 2004). They appear to initially settle on the damaged areas of pink sea fan where the coenenchyma has been damaged and the skeleton exposed (Southward et al., 2004) this would suggest that trap activities may

lead to a greater susceptibility of pink sea fan to fouling organisms, potentially leading to reduced survival over longer timescales.

In abraded *E. verrucosa* colonies, regrowth can occur within days (Readman and Hiscock, 2017); 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 colonisation by epibiota or disease. Subsequent mechanical stress and susceptibility to weakening from the burrowing activities of epibiota could induce mortality (Bavestrello et al., 1997).

Eno et al. (2001) and Tillin et al. (2010) classified fragile sponge and anthozoan communities on subtidal rocky habitats as high sensitivity to surface abrasion. 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. The impacts of abrasion from static gears, such as traps are not well studied but evidence suggests that species such as sponges and soft corals may be left vulnerable to disease and delayed mortality after damage has occurred (Bavestrello et al., 1997; Shester and Micheli, 2011; Readman and Hiscock, 2017).

7.3.2 Removal of target species

The use of traps on and around reef features results in the removal of target species which play a role in maintaining habitat diversity within the ecosystem, including crustacea species such as edible crab (*Cancer pagarus*), European lobster (*Homarus gammarus*), crawfish (*Palinurus elephas*) and other species such as whelks (Gubbay and Knapman, 1999; Sewell and Hiscock, 2005; JNCC and Natural England, 2011). Removal of these species could impact the productivity and community composition of the reef feature as decapods can provide important structure to benthic communities and even regulate trophic cascades in a variety of temperate habitats (including subtidal reefs) (Babcock et al., 1999).

The predation of specific competitors such as mussels or the predation of herbivores by target species such as edible crab and lobster, allows for the maintenance of algal growth (Boudreau and Worm, 2012). However, this is not specific to target species such as edible crab and European lobster, rather as a generalised role of large decapod species on reef features (Tonk and Rozemeijer, 2019). Similar prey preferences have been found for the edible crab and other co-existing crab species indicating niche-competition (Mascaró and Seed, 2001; Griffin et al., 2008; Silva et al., 2008). Therefore, the removal of edible crab through the potting fishery could increase the abundance of co-existing crab species due to reduced competition with edible crab. Silva et al. (2014) reported small-scale migrations of different crab species, indicating that the different species could be interchangeable in terms of ecosystem function and therefore the impact of reduced edible crab abundance on the reef feature is likely to be minimal.

Some evidence shows that inter-specific competition may be the limiting factor for edible crab, rather than removal from fishing. This is evident as increased abundance and size of European lobster, and their predation of the smaller edible crabs can be seen as a result attributed to reduced fishing pressure (Hoskin et al., 2011).

7.3.3 Removal of non-target species

Traps can remove non-target species which could impact community composition. Removal of non-target species includes all incidental catch, such as undersized target species. The mortality of incidental catch in traps is generally considered low due to the selectivity for the target species and high probability of survival for any unwanted species caught and returned (Broadhurst et al., 2006; Suuronen et al., 2012).

A UK-based study by Gall et al. (2020), investigating trap interaction mechanisms and footprints, showed that 14 of 18 taxa identified suffered damage from traps and individuals of six taxa were removed from the reef. Trap hauling damaged or removed between 25% and 30% of observed epibenthic species, with broadly consistent patterns between trap type. Individuals of edible sea urchin (*Echinus esculentus*), black sea urchin (*Holothuria forskali*) and common starfish (*Asterias rubens*) were observed to roll or be moved out of the way by the pressure wave from a trap (Gall et al., 2020). Analysis from this study identified that, following trap haul, significantly more individuals remain undamaged (0.54 individuals per m² (3%)) (Gall et al., 2020). Selected taxa for which more individuals were damaged than were undamaged by inkwell traps were *C. celata* (damaged = 54%, undamaged = 45%), *E. verrucosa* (damaged = 54%, undamaged = 45%), and *P. foliacea* (damaged = 82%, undamaged = 13%) (Gall et al., 2020). 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 (Gall et al., 2020). Consistency was exhibited across trap type in those species groups either impacted or removed from the reef by hauling, and upright species were found to be particularly vulnerable to damage from pots (Gall et al., 2020).

The study noted that while areas fished with static gear 'more closely represent fully functioning benthic rocky reef areas' in comparison to areas that had previously been fished with bottom towed gear, there was not a statistically significant difference between the two (Gall et al., 2020). While recognising that the increased presence of biogenic habitat forming taxa (hydroids, bryozoans and soft corals) in the traps-only areas would ultimately benefit these sites by decreasing sediment movement and water speed, supporting a more biodiverse and productive ecosystem, full recovery of non-target species had not occurred (Gall et al., 2020). It was noted that negative effects of potting may have been compounded by a particularly bad storm season: fishers were unable to retrieve traps that then may have caused greater than usual

damage to species due to longer soak time and increased movement across the seabed (Gall et al., 2020).

Nevertheless, Gall et al. (2020) calls into question earlier short-term studies where no detectable negative effects of potting were found for benthic species (Eno et al., 2001; Coleman et al., 2013; Stephenson et al., 2017), and supports the findings of Rees (2018). Rees (2018) studied impacts of traps over a three-year period using industry standard parlour pots in four 500 x 500 m areas in Lyme Bay at different densities ('no traps' (control) = 0 traps; 'low' = 5 to 10 traps; 'medium' = 15 to 25 traps; 'high' = 30 traps), and found traps had impacts on abundance and species richness, especially in high density areas (Rees, 2018). Two of the indicator species (Ross coral, *P. foliacea* and the ascidian, *Phallusia mammillata*) increased in abundance after three years of reduced trap density, with *P. mammillata* showing positive recovery in the low trap and control treatments and *P. foliacea* showing recovery in control treatments (Rees, 2018).

Both Gall and Rees suggest that while traps may have limited impact on some non-target species, the implications of year-round trapping on the broader reef ecosystem may have been underestimated in short-term studies, and that management approaches should consider potential negative impacts of high spatial and temporal densities of potting effort on the biological functions of reef habitats (Rees, 2018; Gall et al., 2020).

7.4 Variation in impacts

The impacts of traps on rocky reefs will likely depend on several factors, including gear variations, fishing intensity, and habitat and environmental variables. Rocky reefs that are subject to relatively high levels of wave energy, like those in Cape Bank, are likely to be less sensitive to physical impacts than those in more sheltered areas, such as Lyme Bay (Rees, 2018). This is because their ecological components either have higher resilience, resistance, or lower sensitivity.

Studies carried out in Lyme Bay into the impacts of the 2013/2014 storms, on comparable habitat, found reductions in abundance, diversity and richness after the storms, and impacts on selected indicator taxa (Gall et al., 2020). The true impacts of traps may therefore be compounded by impacts from storms (Lewis et al., 2009).

Stephenson et al. (2015) showed that trap movement can occur during spring or neap tides but was more common during spring tide (four of the six days with significant trap movement), indicating that moderate tidal movement may cause abrasion. The distances of trap movements varied with different environmental conditions (swell height and tidal strength), with overall trap movement increasing slightly with increasingly extreme conditions. This suggests that a combination of swell height and tidal strength can affect trap movement. The area potentially impacted by traps moved by swells and tides ranged from 53 to 115 m² per trap (mean = 85.8 m²) but there was no significant difference in area impacted between

neap and spring tide or swell height (Stephenson et al., 2015). At a local scale, trap impacts on epibenthic assemblages and species richness may not be detected against the background of natural variability.

The negative consequences of traps found in the studies above were all concerned with fragile and sensitive species; however, the scale of impact appears to vary between species. Eno et al. (2001) reported some evidence of detachment of ascidians (*Ascidacea*) and sponges (*Porifera*) and damage to Ross coral colonies due to trapping, while sea pens (*Pennatulacea*) were observed reattaching to substrate after smothering or uprooting and pink sea fan (*Eunicella verrucosa*) were observed bending under the weight of a trap but 'springing back' once the trap was removed.

Rees (2018) found that sessile reef species decreased in abundance over time when subjected to long term trapping activity, with Ross coral being particularly sensitive and showing a negative response to even low trap densities. However, prior to the study taking place, a storm removed almost the entire Ross coral population in the area (Rees, 2018). This was therefore a recovering population with low Ross coral abundance, and this may have exacerbated the significance of the trap impacts (Rees, 2018). Pink sea fan abundance decreased in both the medium and high trap intensity treatments, but decreases were not statistically significant (Rees, 2018). However, it was noted that this may indicate a longer-term response that their three-year period did not completely document (Rees, 2018).

Rees (2018) challenges the perception that commercial traps are benign based on more recent observations at 'normal' trap levels (i.e. set/haul two to three times a week in stable weather and once a week in unstable weather). Many recovering species, subsets and selected indicator species showed no impact from above-normal levels of traps on relative abundance, species richness and assemblage composition; however, Ross coral (*P. foliacea*) under low, medium and high trap treatments, and the sea squirt (*P. mammillata*) under medium and high treatments showed impacts of trap activity on recovery after three years (Rees, 2018).

Although Stephenson et al. (2017) focussed on algal dominated reef features which can be less sensitive, the study found that no declines in abundances of erect species were found under intensive trap fishing, supporting the findings of previous UK studies (Eno et al., 2001; Coleman et al., 2013). Direct impacts were not observed in intensively or lightly fished rocky reef habitats (specifically, 'faunal and algal crusts on exposed to moderately wave-exposed circalittoral rock') (Stephenson et al., 2017). A shift in community composition was found but attributed to natural change when controls were considered (Stephenson et al., 2017).

The physical footprints of traps are much smaller than mobile gears such as trawls and dredges (Nielsen et al., 2013), making it very unlikely that traps fished commercially would land, soak and be hauled, in exactly the same location on successive fishing trips (Eno et al., 2001). Erect species typically found in the

habitats studied (such as *Laminaria hyperborea*, red and brown algae >1 cm, *Delessaria sanguinea*, *Flustra foliacea*, *Alcyonium digitatum* and feather hydroids (for example *Nemertesia ramosa*)) are all expected to recover between 6 and 36 months from mechanical interference (based on crushing, physical blows, or rubbing and erosion of a single passage of a standard scallop dredge across the organism (Jackson, 2004; Tyler-Walters, 2006; Tyler-Walters and Ballerstedt, 2007; Budd, 2008). It is likely, therefore, that these species will not be severely damaged and are expected to recover between fishing events. It is noted that other species, for example axinellids, may take up to 25 years to recover and structurally complex species such as, *Lithothamnion coralloides*, much longer (Stephenson et al., 2017). It should be noted, however, that this study only looked at trap activities from July to September in 2014 and July and August in 2015, despite traps being used all year round, and study sites were small (25 x 10 m and 5 x 10 m control areas) (Stephenson et al., 2017).

7.5 Summary of the effects of traps on rocky reef

The majority of literature before 2015 has suggested that traps are unlikely to significantly impact rocky reef biotopes. However, more recent studies, such as those conducted by Gall et al. (2020) and Rees (2018) suggest that traps will have negative impacts on the biological functions of reef habitats at high spatial and temporal densities. A site level assessment considering the site conservation objectives, intensity of fishing taking place, exposure to natural disturbance and potential presence of particularly sensitive species will be needed to determine whether management will be required.

The site level assessments will assess fishing activities for their impact upon protected habitats and species (in this case, the relevant biotopes for rocky reef). Specifically, this assessment considers the potential for these activities to hinder the conservation objectives of the MCZ or have an adverse effect on the site integrity of the SAC. The data used in the assessment will include VMS data, as well as feature habitat data from JNCC and Natural England. Where the assessment concludes that the current level of management is not sufficient to protect the designated features of the site, recommended management options will be provided. MMO has regard to the best available evidence and through consultation with relevant advisors, stakeholders, and the public, will conclude which management option is implemented.

Site level assessments may conclude that management of the interaction between traps and rocky reef biotopes may be unnecessary for MPAs designated for these features. In which case, a site monitoring and control plan, including regular monitoring of this fishing activity with no restrictions, may be suggested to be sufficient at this stage.

8 Biogenic reef (*Sabellaria* spp.)

This section brings together and analyses the available evidence on how traps affect biogenic reef features.

Reefs are an Annex I habitat listed in the Council Directive 92/43/EEC (the Habitats Directive). Several MCZ features including circalittoral and infralittoral rock, subtidal chalk, and Ross and honeycomb worm reefs correspond to the Annex I reef classification. JNCC classifies reef into one (or more) of the following three subtypes: bedrock, stony and biogenic (Duncan et al., 2022). For the purpose of these literature reviews, MMO has separated reefs into two different categories; bedrock reef and stony reef are categorised as ‘rocky reef’ and biogenic reef is categorised as ‘biogenic reef (*Sabellaria* spp.)’. This section only refers to biogenic reef (*Sabellaria* spp.).

Although other biogenic reef habitats exist such as mussel (*Mytilus edulis*) beds, this document only considers reefs formed by the two different species of *Sabellaria* worm (*Sabellaria* spp.) as these are the only types of biogenic reef found within the relevant sections of the MPAs, where MMO is the principal regulator for fishing. These sites are Goodwin Sands (Natural England, 2021) which contains ross worm reef (*Sabellaria spinulosa*); Haisborough, Hammond and Winterton which contains ross worm reef (*Sabellaria spinulosa*) (JNCC, 2018e); Inner Dowsing, Race Bank and North Ridge which contains ross worm reef (*Sabellaria spinulosa*) (JNCC, 2018f); and North Norfolk Sandbanks and Saturn Reef which contains ross worm reef (*Sabellaria spinulosa*) (JNCC, 2017b).

8.1 Overview of the sensitivity of biogenic reef to traps

8.1.1 Sensitivity – resistance to damage

When conditions are favourable, dense aggregations of *Sabellaria* spp. form reefs. These reefs are structurally fragile and therefore interactions with fishing gear have the potential to negatively impact the habitat and associated biotopes. They are most sensitive to substratum loss and displacement as the worms are fixed to the substratum and cannot reattach once dislodged or rebuild their tubes if removed from them (OSPAR Commission, 2010). *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 (for example a subtidal reef is unlikely to be exposed to trampling, however at spring tides this may occur if the reef is exposed). Sensitivity assessments of *Sabellaria* spp. reefs identify the main pressures from fishing activities to be abrasion/disturbance of the substrate on the surface of the seabed and removal of non-target species (Jackson and Hiscock, 2008).

The sensitivity of *Sabellaria* spp. reefs to different pressures has been assessed following the MarESA approach (Tyler-Walters et al., 2018). Individual biotope sensitivities range from low to high. These are then used in JNCC’s and Natural

England's AoOs to determine the site level sensitivity of the designated habitat feature to various pressures. JNCC and Natural England (JNCC and Natural England, 2011) report the sensitivity of *Sabellaria spp.* reefs as medium, depending on fishing intensity, while others 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.

Sensitivity of *Sabellaria spp.* reef to, and recovery from, fishing activity will depend on several factors including environmental conditions, which particular *Sabellaria spp.* is present, the sensitivity of that species, and level of exposure to the pressures/disturbance.

8.1.2 Recovery – rate of recovery

Certain disturbance events such as fracturing damage, or partial removal of *Sabellaria spp.* reef structure, may not always result in the disappearance of the reef. Evidence has shown that damaged parts of the reef can be rebuilt in time, depending on the extent and nature of the damage; this could be as quickly as within a few days, suggesting high recovery of the species (Salomidi et al., 2012). It has been illustrated that *Sabellaria spp.* polychaetes release gametes when removed from their tubes (Pearce et al., 2011). This spawning response increases their resilience to disturbance and in some cases means the disturbance can enhance reef structure (Pearce et al., 2011). However, recovery is dependent on the supply of suitable material with which to repair the damaged part of the tube, as such, a lack of material could result in further erosion of the reef (Last et al., 2011). Although disturbance can potentially result in enhancement of reef structure and high recovery rates, there is an initial immediate impact to reef communities which could have longer term impacts on the communities' recovery (Salomidi et al., 2012). Although there is no evidence which quantifies the recovery rate from fishing disturbances, Jones et al. (2000) suggests that *S. spinulosa* could recolonise after winter storm damage up to 2.4 cm by the following summer. Recovery rates will also vary depending on several factors, such as season of impact, larval supply, recruitment, and local environmental factors (Gibb et al., 2014; Tillin et al., 2015). A report by Gibb et al. (2014), extrapolating results of recovery from post beam trawl studies of *S. alveolata* and applying them to *S. spinulosa*, predicts that recovery through repair and larval re-colonisation could occur within 2 -10 years, if sufficient proportions of reef and worms survive.

8.2 Level of literature, caveats and assumptions

Walmsley et al. (2015) have noted that there is no primary evidence on the direct impact of traps on *Sabellaria spp.* reef. However, sensitivity assessments based on expert knowledge are available. Consequently, this review uses both direct peer reviewed evidence and grey literature (for example, reports, policy literature and government documents) to review the impacts of traps on biogenic reefs. Key

evidence gaps are around trap intensity and location, long time series data and natural variability (Walmsley et al., 2015). While there has been more recent interest, and sensitivity assessments do exist (Hall et al., 2008), there is still a lack of evidence. As such, this review has used literature for both *Sabellaria spp.* (*S. spinulosa* and *S. alveolata*), as this is the best available evidence, although it is recognised that both species will have different sensitivities.

The review produced by Gibb et al. (2014) illustrates that there is limited evidence on recovery rates, stability, and persistence of *Sabellaria spp.* reef as a result of subtidal trap activity. Gibb et al. (2014) does illustrate how lighter damage of *Sabellaria spp.* reef results in quicker recovery of around 4-5 days, compared to longer periods of time from heavy damage causing large cracks in *Sabellaria spp.* reef, especially if reef damage is further exacerbated by wave action. Gibb et al. (2014) showed that there is some evidence where intertidal *S. alveolata* reef took 23 days to recover from severe damage after trampling. It is recognised that some caution is needed in extending this conclusion to *S. spinulosa* and that *S. spinulosa* have been recorded to be more fragile and less resilient than *S. alveolata* reefs, (Cunningham et al., 1984; Gibb et al., 2014).

A review by Rees (2018) concluded that there is a scarcity of empirical studies on the impact of traps on reefs, and that the evidence is varied with some studies finding little evidence of impacts (Coleman et al., 2013; Haynes et al., 2014; Stephenson et al., 2015; Rees, 2018). Rees (2018) states that due to the short timescales of the studies it is not possible to eliminate the potential for cumulative or longer-term impacts. However, it should be noted that the impacts discussed in more detail are for a number of species and not just *Sabellaria spp.* Abrasion caused by pots will also vary as pots themselves may have different construction materials and differ in dimensions and weight (Walmsley et al., 2015).

8.3 The pressures of traps on biogenic reef

As a result of traps, this feature may be sensitive to the following pressures, so they are considered in this document:

- abrasion or disturbance of the substrate on the surface of the seabed
- removal of non-target species.

There is insufficient evidence available to determine whether this feature is sensitive to the following pressures as a result of the use of traps:

- hydrocarbon and PAH contamination
- introduction of light
- litter
- organic enrichment.

8.3.1 Abrasion/disturbance of the substrate on the surface of the seabed

Traps can damage reef (including *Sabellaria spp.*) through gear striking or becoming entangled with the reef. This is most likely on deployment, through movement of gear on the benthos due to tide, current and storm activity, and as the gear is dragged along the seafloor on retrieval (Coleman et al., 2013; Grieve et al., 2014; Gall et al., 2020). The use of traps is generally considered far less damaging to benthic habitats than the use of mobile gears (Sewell and Hiscock, 2005).

Physical disturbance can result in epifauna, especially emergent species such as erect sponges and coral being dislodged or damaged when snagged (on a trap for example), although there are limited studies of such effects (Gubbay and Knapman, 1999; Sewell and Hiscock, 2005; Polet and Depestele, 2010; Lart, 2012; Coleman et al., 2013; Walmsley et al., 2015).

Gibb et al. (2014) state that abrasion on the surface of *S. spinulosa* reef is likely to damage the ends of the worm tubes and may cause greater damage where areas are broken apart. Although Sewell and Hiscock (2005) suggest there is little likelihood of damage from traps on *S. spinulosa* reef, they also state that erect *S. spinulosa* reef communities can be physically damaged by traps when deployed, dragged or extracted .

Biogenic reef on sediment habitats, listed in Hall et al. (2008), refer to *S. alveolata* and have been described as having medium sensitivity to heavy levels of trap fishing. *S. spinulosa* reefs have been described as having a medium sensitivity to abrasion (Tillin et al., 2020) and are known to be more sensitive to the impacts of abrasion than *S. alveolata* (Gibb et al., 2014).

Traps cause direct physical impacts to biogenic reef which may have biological implications for the polychaetes forming the reef and the flora and fauna associated with the reef. Traps cause damage to the reef structure, reducing the substrate available for species to attach to (Kaiser et al., 2006). Damage and fragmentation of reef also reduces the habitat structure and complexity, reducing its capacity to support epifauna and infauna communities (JNCC, 2011; Last et al., 2011), although these effects are less significant than for trawling. Gibb et al. (2014) cites studies which show *S. alveolata* reefs recovered within 23 days from trampling, walking and stamping (Cunningham et al., 1984). However, Cunningham et al. (1984) also reported that more severe damage caused by kicking and jumping on the reef was still not fully repaired 23 days later. Anchor contact and dragging could cause similar levels of damage but it is unclear how reefs would recover. *S. spinulosa* reefs are also recorded to be more fragile and less resilient than *S. alveolata* reefs, meaning the impacts of abrasion/disturbance may be greater and recovery times longer (Gibb et al., 2014) than those observed in *S. alveolata* (Cunningham et al., 1984).

Sensitivity to abrasion can modify *Sabellaria spp.* reef and associated communities (Salomidi et al., 2012). Impacts are greater from trawling than from traps (Bouma et

al., 2009). The physical footprints of traps are much smaller than mobile gears such as trawls and dredges (Nielsen et al., 2013), and it is unlikely that they would land, soak and be hauled, in exactly the same location on successive fishing trips (Eno et al., 2001). The potential contact area for traps is significantly larger than just the contact the trap has with the seabed due to the need to account for the distance travelled by pots moving with currents and tides (for example for inkwell pots the total contact area could be 7.16 m² compared to a seabed contact area of 3.51 m²) (Natural England, 2022a). Any loss of reef structure can result in reduced species abundance/richness, biomass, and consequentially ecosystem functioning (Salomidi et al., 2012). Although *Sabellaria spp.* have medium resilience to abrasion (Tillin et al., 2015), local environmental factors such as current strength and sediment supply may increase impacts felt by traps (Salomidi et al., 2012).

8.3.2 Removal of non-target species

Traps have the potential to remove non-target species (bycatch) which can impact species abundance, community composition and food web interactions (Alverson et al., 1994; Bullimore et al., 2001; Gibb et al., 2014). For example, the removal of species such as juvenile fish that prey on *S. spinulosa* worms could be beneficial to *S. spinulosa* due to reduced predation but may be detrimental to the overall reef feature, which includes the reef's associated communities. Gibb et al. (2014) cites previous studies which show non-commercial butterflyfish (*Pholis gunnellus*) and dragonet (*Callionymus lyra*) prey on *S. spinulosa* worms. The common shore crab (*Carcinus maenas*) is also known to predate *S. spinulosa* (Taylor et al., 1962; Gibb et al., 2014). The brittlestar (*Ophiothrix fragilis*) which can form dense aggregations, also competes with *S. spinulosa* worms for food and space. However, there is no empirical evidence to demonstrate how removal of these predators impact *S. spinulosa* worms or the reef feature as a whole (Gibb et al., 2014).

The mortality of incidental catch in traps is considered low due to selectivity of target species and high probability of survival for any bycatch returned (Broadhurst et al., 2006; Suuronen et al., 2012). Gibb et al. (2014) reported that evidence for the ecological interaction between *Sabellaria spp.* and other species was limited, but that there was no evidence for significant biological effects on *S. spinulosa* reef related to removal of non-target species. Despite the possible positive effects discussed above that the removal of non-target species pressure could have on *Sabellaria spp.* worms in relation to reduced predation and competition for food and space, *Sabellaria spp.* reefs are considered to be sensitive to the pressure. Removal of non-target species may impact the associated biological communities of the reef feature. It is also essential to note that the biogenic structure created by *Sabellaria spp.* worms is the key characterising feature of the biotope, therefore removal of the worms and tubes as bycatch would remove the biotope (Tillin et al., 2015).

8.4 Variation in impacts

The impacts of traps on *Sabellaria spp.* reef depends on several factors, such as gear type, fishing intensity, habitat, and environmental variables. Much of the available information on sensitivity is based on trawling or dredging (d'Avack et al., 2014; Walmsley et al., 2015) and so a conclusion directly relating to traps is difficult to determine.

S. spinulosa has been presented as having a medium sensitivity to all levels of traps, due to its robust nature and ability to recover quickly, depending on the extent of damage (d'Avack et al., 2014; Walmsley et al., 2015). Others have considered sensitivity to be minimal (Holt et al., 1998). Tillin et al. (2010) concluded sensitivity for both *S. alveolata* and *S. spinulosa spp.* to be low in relation to the abrasion pressure from static gear such as traps. JNCC and Natural England (2011) also suggest sensitivity of *S. spinulosa* reef to static gear such as traps is medium but state that this depends on fishing intensity. Overall, sensitivity of *Sabellaria spp.* to traps is considered medium, subject to levels of effort and environmental variables affecting the severity of impact. The individual impact of a single fishing operation may be slight but cumulative damage may be significant (Eno et al., 2001; Foden et al., 2010; Roberts et al., 2010). Walmsley et al. (2015) suggest that impacts will vary depending on a site's environmental characteristics, but it should also be noted that *Sabellaria spp.* reefs have a standard preferred substrate of coarse/mixed sediment (Jackson and Hiscock, 2008). Although traps have been concluded to have a low impact by authors such as Walmsley et al. (2015), this relates to the *S. spinulosa* reef's ability to recover from minor disturbance. This is only possible provided existing reef remains and environmental conditions are stable. The 'low' impact rating also relates to low intensity, frequency, and footprint and is based on trawling-based sensitivity assessments. As such, there is the potential for an increase in intensity, frequency and/or footprint to increase the level of impact. Although, not specifically for *Sabellaria spp.* reefs, a study by Rees (2018) of trapping activities in Lyme Bay (UK) found that damage on rocky reefs was only significant if the intensity of fishing was high enough (defined by the author as ~30 pots per 500 m²). The impact will also vary with the type of trap being used, for example literature noted that the total potential contact area (total distance travelled x area of the trap base) for inkwell pots was 7.16 m² compared to parlour pots with 5.24 m² ± 0.67 (Natural England, 2022a).

8.5 Summary of the effects of traps on biogenic reef

The best available evidence shows that traps have the potential to adversely affect *Sabellaria spp.* reefs. However, studies are limited, with some noting little evidence of impacts (Coleman et al., 2013; Haynes et al., 2014; Stephenson et al., 2015; Rees, 2018). Local conditions (such as strength of sediment supply and currents) and the intensity of activities may change the level of impact and recoverability (Salomidi et al., 2012). Therefore, a site level assessment considering the site

conservation objectives, intensity of fishing activity taking place, exposure to natural disturbance and potential presence of particularly sensitive species will be needed to determine whether management will be required.

The site level assessment will assess fishing activities for their impact upon protected habitats and species. Specifically, this assessment considers the potential for these activities to hinder the conservation objectives of the MCZ or have an adverse effect on the site integrity of the SAC. The data used in the assessment will include VMS data, as well as feature habitat data from JNCC and Natural England. Where the assessment concludes that the current levels of management is not sufficient to protect the designated features of the site, recommended management options will be provided. MMO has regard to the best available evidence and through consultation with relevant advisors, stakeholders, and the public, will conclude which management option is implemented.

9 Annex I sandbanks which are slightly covered by sea water all the time and MCZ subtidal sediment habitats

This section brings together and analyses the available evidence on how traps affect Annex I sandbanks which are slightly covered by sea water all the time and marine conservation zone subtidal sediment habitats (hereafter referred to as sandbanks and sediments).

Sandbanks which are slightly covered by sea water all the time (hereafter referred to as sandbanks) are an Annex I habitat listed in Council Directive 92/43/EEC (the Habitats Directive). They are a designated feature of the SACs listed in Table 2. Sandbanks can be further classified into EUNIS habitat types. With the exception of subtidal mud, which is not found upon sandbanks, these EUNIS habitats correspond with MCZ subtidal sediment broadscale habitats. MCZ subtidal sediment habitats are designated features of the MCZs listed in Table 2.

Table 2. MPAs containing designated features of Annex I sandbanks or relevant MCZ broadscale habitats.

Bioregion	Relevant MPA	Relevant Features				
		Annex I sandbanks which are slightly covered by sea water all the time	Subtidal coarse sediment	Subtidal mixed sediments	Subtidal sand	Subtidal mud
Eastern Channel	Albert Field MCZ		X	X		
	Bassurelle Sandbank SAC	X				
	East of Start Point MCZ				X	
	Foreland MCZ		X		X	
	Goodwin Sands MCZ		X		X	
	Inner Bank MCZ		X	X	X	
	Offshore Brighton MCZ		X	X		
	Offshore Overfalls MCZ		X	X	X	
West of Wight-Barfleur MCZ		X	X			
Irish Sea	Fylde MCZ				X	X
	Shell Flat and Lune Deep SAC	X				
	West of Copeland MCZ		X	X	X	
	West of Walney MCZ				X	X
Northern North Sea	Farnes East MCZ		X	X	X	X
	Fulmar MCZ			X	X	X
	North East of Farnes Deep MCZ		X	X	X	X
	Swallow Sand MCZ		X		X	
Southern North Sea	Haisborough, Hammond and Winterton SAC	X				
	Holderness Offshore MCZ		X	X	X	
	Kentish Knock (East) MCZ		X	X	X	
	Margate and Long Sands SAC	X				
	Markham's Triangle MCZ		X	X	X	X
	North Norfolk Sandbanks and Saturn Reef SAC	X				

Bioregion	Relevant MPA	Relevant Features				
		Annex I sandbanks which are slightly covered by sea water all the time	Subtidal coarse sediment	Subtidal mixed sediments	Subtidal sand	Subtidal mud
	Orford Inshore MCZ			X		
Western Channel and Celtic Sea	Cape Bank MCZ		X			
	East of Haig Fras MCZ		X	X	X	X
	Greater Haig Fras MCZ		X	X	X	X
	Hartland Point to Tintagel MCZ		X		X	
	North East of Haig Fras MCZ		X		X	X
	North West of Jones Bank MCZ		X	X	X	X
	North West of Lundy MCZ		X			
	South of Celtic Deep MCZ		X	X	X	
	South of the Isles of Scilly MCZ		X	X	X	
	South West Approaches to Bristol Channel MCZ		X		X	
	South West Deeps (East) MCZ		X		X	
	South West Deeps (West) MCZ		X	X	X	X
	Western Channel MCZ		X		X	

9.1 Feature summaries

9.1.1 Sandbanks

Sandbanks consist of sandy sediments that are permanently covered by shallow sea water, typically at depths of less than 20 m below chart datum. The habitat comprises distinct banks which may arise from horizontal or sloping plains of sandy sediment.

The diversity and types of community associated with this habitat are determined particularly by sediment type together with a variety of other physical, chemical, and hydrographic factors.

Within the UK's offshore waters, sediments can be categorised into a number of EUNIS habitat types as follows:

Subtidal coarse sediment

Coarse sediments include coarse sand, gravel, pebbles, shingle, and cobbles which are often unstable due to tidal currents and/or wave action. These habitats are generally found on the open coast or in tide-swept channels of marine inlets. They typically have a low silt content and a lack of a significant seaweed component. They are characterised by a robust fauna including venerid bivalves (EEA, 2019a).

Subtidal sand

Subtidal sands consist of clean medium to fine sands or non-cohesive slightly muddy sands which are most commonly found on open coasts, offshore or in estuaries and marine inlets. Such habitats are often subject to a degree of wave action or tidal currents which restrict the silt and clay content to less than 15%. This habitat is characterised by a range of taxa including polychaetes, bivalve molluscs and amphipod crustacea (EEA, 2019b).

Subtidal mud

Subtidal mud and cohesive sandy mud are found in marine areas extending from the extreme lower shore to offshore, circalittoral habitats. Unlike the subtidal sand, coarse and mixed sediments, subtidal mud does not occur on sandbanks. This biotope is predominantly found in sheltered harbours, sea lochs, bays, marine inlets and estuaries and stable deeper/offshore areas where the reduced influence of wave action and/or tidal streams allow fine sediments to settle. Such habitats are often dominated by polychaetes and echinoderms, in particular brittlestars (such as *Amphiura* spp.). Estuarine muds tend to be characterised by infaunal polychaetes and oligochaetes. Sea pens such as *Virgularia mirabilis* and burrowing megafauna (including *Nephtys norvegicus*) communities (SPBMFC) are common in deeper muds and are also an MCZ habitat of conservation importance (HOCl). This specific HOCl has been assessed separately, see section 4 (EEA, 2019c).

Subtidal mixed sediments

Subtidal mixed sediments are found from the extreme low water mark to deep offshore circalittoral habitats. These habitats incorporate a range of sediments including heterogeneous muddy gravelly sands and mosaics of cobbles and pebbles embedded in or lying upon sand, gravel, or mud. There is a degree of confusion with regards to nomenclature within this complex as many habitats could be defined as containing mixed sediments, in part depending on the scale of the survey and the sampling method employed. The British Geological Survey trigon (see: Figure 5 in McBree et al., 2011) can be used to define truly mixed or heterogeneous sites with surficial sediments which are a mixture of mud, gravel and sand. However, another 'form' of mixed sediment includes mosaic habitats such as superficial waves or ribbons of sand on a gravel bed or areas of lag deposits with cobbles/pebbles embedded in sand or mud and these are less well defined and may overlap into other habitat or biological subtypes. These habitats may support a wide range of infauna and epibiota including polychaetes, bivalves, echinoderms, anemones, hydroids and Bryozoa. Mixed sediments with biogenic reefs or macrophyte dominated communities are classified separately. Subtidal biogenic reefs are assessed separately under section 8. No MPAs currently being assessed by MMO are designated to protect subtidal macrophyte-dominated sediments, however they do represent supporting habitats for marine birds within special areas of protection (SPAs) (EEA, 2019d, 2019e, 2019f).

9.1.2 Supporting habitats

As well as being designated MPA habitats requiring protection in their own right, subtidal sediments act as important supporting habitats for other MPA designated features. These include MCZ species such as ocean quahog, sea pens and fan mussel. The dedicated review sections provide further detail on the specific supporting habitat(s) for each protected feature, see sections 4 to 9.

With regard to MCZ features, supporting sedimentary habitats can provide the substrate for the benthic communities to grow and thrive, supporting ecological processes and the wider food web. The potential impact of fishing gears on the supporting substrate is discussed within this sandbank and subtidal sediment review. The potential impact of fishing gears on the MCZ features themselves is discussed in their dedicated sections.

9.2 Overview of the sensitivity of sandbanks and sediments to traps

9.2.1 Sensitivity – resistance to damage

Sandbanks and subtidal sediments are less sensitive and likely to recover more quickly from fishing activity impacts than more fragile habitats such as biogenic reefs, however fishing activity still has the potential to negatively impact these

habitats and hinder the conservation objectives of the sites in which they are protected, particularly with regard to the structure and function of the biological communities present. This is especially true in intensively fished areas which are likely to be maintained in a permanently altered state, inhabited by fauna adapted to frequent physical disturbance due to the inability of the habitat to sufficiently recover before the next passing of fishing gear (Collie et al., 2000).

The sensitivity of sandbank and sediment features to traps is similar to that of nets and lines where surface abrasion and disturbance could be caused during setting and retrieval (Roberts et al., 2010). Sensitivity to, and their recovery from, fishing activity will depend on several factors including the sediment type, presence of particularly sensitive species, exposure to natural disturbance (Natural England, 2022a), as well as recruitment of new individuals (Collie et al., 2000), growth of surviving biota, and active immigration from adjacent habitat (Brey, 1999).

9.2.2 Recovery – rate of recovery

Clean sand communities are likely to recover from disturbance most quickly (Collie et al., 2000), whereas communities from gravel (subtidal coarse sediment) and muddy sand habitats tend to have the slowest physical and biological recovery rates (Dernie et al., 2003; Kaiser et al., 2006; Foden et al., 2010). When considered in terms of MCZ subtidal sediment habitats, muddy sand and clean sand habitats would both fall under the subtidal sand classification which highlights the complexity of understanding the impacts of fishing impacts on sedimentary habitats. Little evidence is available regarding the sensitivity and recovery of subtidal mixed sediments but in general terms the more physically stable habitats are, such as subtidal mud and coarse sediments like gravel, the longer recovery is likely to take (Collie et al., 2000).

9.3 Level of literature, caveats and assumptions

This literature 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 studies undertaken elsewhere. There is limited evidence of the impacts of traps on sandbanks, sediments and their biological communities, however they are unlikely to significantly impact the physical structure of the feature and biological impacts are not likely to be a concern unless activities reach a particularly high level of intensity, or particularly sensitive species are present (Hall et al., 2008; Roberts et al., 2010; Tillin et al., 2010; JNCC and NE, 2011; Walmsley et al., 2015).

Abrasion caused by traps will vary as traps themselves may have different construction materials and differ in dimensions and weight (Walmsley et al., 2015). The use of traps is generally considered far less damaging to benthic habitats than the use of mobile gears (Sewell and Hiscock, 2005).

9.4 The pressures of traps on sandbanks and sediments

As a result of traps, these features may be sensitive to the following pressures, so they are considered in this document:

- abrasion or disturbance of the substrate on the surface of the seabed
- removal of target species
- removal of non-target species.

There is insufficient evidence available to determine whether this feature is sensitive to the following pressures as a result of the use of traps:

- hydrocarbon and PAH contamination
- introduction of light
- litter
- synthetic compound contamination
- transition elements and organo-metal contamination

9.4.1 Abrasion/disturbance of the substrate on the surface of the seabed

There is little primary evidence on the impact of traps on sediment habitats (Walmsley et al., 2015). However, available evidence suggests that static gears have a relatively low impact on benthic communities in comparison to towed gears, as a result of the small footprint of the seabed affected (Roberts et al., 2010). On average, traps targeting crustacea such as crab and lobster are in the region of 50 x 70 cm (MRAG Ltd and Envision Mapping Ltd, 2015), however, the number of traps laid will vary per site and with the size of vessels operating in the site due to their capacity to carry large strings of traps on board.

Static gears such as traps are unlikely to significantly impact the physical structure of the sandbanks or sediment habitats. There is however potential for impacts to biological communities.

Traps are not likely to be a concern unless it reaches a particularly high level of intensity, or particularly sensitive species are present, as there is the potential for the snagging of gear and subsequent entanglement and damage to fragile epifauna as the level of fishing activity and therefore density level of anchors and ropes increases (Hall et al., 2008; Roberts et al., 2010; Tillin et al., 2010; JNCC and NE, 2011; Walmsley et al., 2015).

Abrasion from traps is possible through the interaction between the seabed and the gear itself including associated lines and anchors. This is more likely to occur if the gear moves across the seabed during hauling of gear or when the gear is subject to strong tides, currents, or storm activity. A study by Gall et al. (2020) on inkwell and parlour pots during hauling found that actual seabed contact was roughly half of the total possible contact area, with pots not consistently dragging along the seabed as might be assumed.

Sensitivity of erect epifauna to potting activity is species-dependent (Roberts et al., 2010). Sea fans and sea pens have been shown to be able to recover from impacts caused by traps, by bending (sea fans) and reinserting themselves following uprooting (sea pens) (Eno et al., 2001). Reinsertion of undamaged sea pens appears rapid, with some species of sea pens recovering and reinserting themselves from uprooting within 72 hours (Eno et al., 2001). Eno et al. (2001) observed no lasting effects on the substrate during their study but did note that whilst sea pens righted themselves after traps were removed, it remains unknown whether they would suffer from potential long-term effects if repeatedly uprooted.

9.4.2 Removal of target species

Traps used in sediment habitats tend to target a range of demersal species such as lobster, crab, shrimp, whelks, and some fish species. Traps directly result in the removal of target species which will play a role in maintaining habitat diversity within the ecosystem, however these species do not tend to be considered 'key and influential' species (species that play a critical role in maintaining the structure and function of the protected feature) nor do they tend to be considered part of a 'characteristic community' (which includes representative communities, such as those covering large areas, and notable communities, such as those that are nationally or locally rare or are particularly sensitive). As such the presence of these target species within sites is unlikely to be linked to the achievement of the conservation objectives of MPAs and management measures are unlikely to be required to limit the impact of this pressure via traps. However, site-specific assessments are required to confirm this.

9.4.3 Removal of non-target species

Bycatch of non-target species from crab and lobster traps around the UK is low. A Marine Stewardship Council report found that only 1% of total catch (excluding undersize and berried individuals returned to the sea before landing) was made up of bycatch in the crab potting fishery around the Shetland Islands (House et al., 2018). Very little bycatch is expected from traps as the design means that fish and shellfish can escape easily before the gear is hauled. Any bycatch can also be released back into the sea immediately without harm.

9.5 Variation in impacts

As with other gears, the likely impact of traps on sandbank and sediment features will vary based on several factors including exposure to natural disturbance, intensity of activity and the sub-features and species which make up the habitats. As discussed previously the sensitivity of erect epifauna to potting activities is species-dependent (Roberts et al., 2010).

Subtidal coarse sediment

Subtidal coarse sediments can have a wide range of sensitivities (JNCC and Natural England, 2011). Communities on unstable coarse sediments are considered to contain relatively robust fauna which are not believed to be greatly impacted by surface abrasion (Hall et al., 2008). More stable gravels may support a 'turf' of fragile species which are more easily damaged (Stewart and Howarth, 2016). Subtidal coarse sediments are generally not considered to be sensitive to static demersal gears such as traps (Tillin et al., 2010; JNCC and Natural England, 2011). However, there is the potential for the snagging of gear and subsequent entanglement and damage to fragile epifauna, particularly as the level of fishing activity and thereby density level of anchors and ropes increases (Walmsley et al., 2015).

Subtidal sand

No primary evidence is available on the impact of potting on subtidal sand. However, sensitivity assessments indicate that the impact of traps is of limited concern due to the generally high energy environments where subtidal sand occurs and the likely greater impact of natural disturbance in these environments compared with potting. The exception to this however may be sand habitats with high levels of potting intensity and presence of sensitive, erect epifauna (Roberts et al., 2010) because of the potential impacts on the biological community should sufficient numbers of sensitive species be damaged or removed.

Subtidal mixed sediments

No primary evidence is available on the impact of potting on subtidal mixed sediments, however, preliminary indications from a study concerning effects of potting on sandbanks suggest there is no detectable effect (Walmsley et al., 2015). As per other habitats, traps are not likely to have a significant impact albeit with the caveats discussed previously regarding high intensity of potting activity, and the increased potential for snagging and entanglement of fragile epifauna (Hall et al., 2008; Roberts et al., 2010). Sensitivity assessments indicate that stable species in rich mixed sediment habitats, as well as oyster beds occurring in these sediments, have low sensitivity to potting levels except heavy levels where sensitivity becomes medium (Hall et al., 2008). This was corroborated by Tillin et al. (2010) where subtidal mixed muddy sediments, sheltered muddy gravels and *Ostrea edulis* beds were suggested to have medium sensitivity to surface abrasion.

Subtidal mud

Primary evidence is available for potting impacts on subtidal mud from two experimental studies concerning sea pens (Eno et al., 2001; Adey, 2007). The studies used sea pens as an indicator of physical disturbance and found impacts from traps were low with no lasting effects on the muddy substrate (Eno et al., 2001; Adey, 2007). Eno et al. (2001) did observe varying responses of different sea pen species to traps suggesting some species may be more tolerant to disturbance than

others based on their ability to withdraw into the sediment and flex under the pot. Some sea pens were also able to reinsert themselves into the sediment following uprooting (Eno et al., 2001). However, no assessment of long-term effects was undertaken by Eno et al. (2001).

Adey (2007) observed traps can remove moderate quantities of sea pens. However, within trap grounds high densities of sea pens can still occur, indicating minimal overall impact at a habitat scale. Tillin et al. (2010) considered that burrowed mud, subtidal mixed muddy sediments and sea pen and burrowing megafauna communities have medium sensitivity to surface abrasion.

As per other sediment types, sensitivity assessments suggest traps are of limited concern on subtidal muds, due to their limited contact with the seabed (Hall et al., 2008; Roberts et al., 2010; JNCC and NE, 2011). Albeit with the same caveat for potential snagging of gear and subsequent entanglement and damage to fragile epifauna, particularly as the level of fishing activity and therefore density level of traps increases (Hall et al., 2008).

9.6 Summary of the effects of traps on sandbanks and sediments

Traps are unlikely to adversely affect Annex I sandbank features, associated sediment sub-features or pose a significant risk of hindering the conservation objectives for MCZ sediment features, meaning that management will likely be unnecessary for Stage 3 SACs and MCZs designated for these features. However, a site-specific assessment considering the site conservation objectives, intensity of fishing activity taking place, exposure to natural disturbance and potential presence of particularly sensitive species will be needed to determine whether management will be required.

The site-specific assessment will assess fishing activities for their impact upon protected habitats and species. Specifically, this assessment considers the potential for these activities to hinder the conservation objectives of the MCZ or have an adverse effect on the site integrity of the SAC. The data used in the assessment will include VMS data, as well as feature habitat data from JNCC and Natural England. Where the assessment concludes that the current level of management is not sufficient to protect the designated features of the site, recommended management options will be provided. MMO has regard to the best available evidence and through consultation with relevant advisors, stakeholders, and the public, will conclude which management option is implemented.

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Annex 1 Gear pressures on sensitive features - traps

This annex summarises the pressures of traps on the features described in this document.

JNCC and Natural England's advice on operations (AoO) provide generic information on pressures that may be exerted by all marine industries, they are an evidence-based product to be used to guide assessments together with bespoke advice from JNCC and Natural England. This is explained further in [Natural England's conservation advice guidance](#).

The sensitivities of designated features to gear pressures were derived using a staged approach. JNCC and Natural England's conservation advice packages (CAP) and AoO have been used by MMO to determine the sensitivities of each feature to the potential pressures from traps, based on actual or representative sites to highlight subject areas for evidence gathering. JNCC and Natural England also provided additional guidance about pressure/feature interactions that should be considered.

An evidence-gathering activity was then carried out. Evidence gathering and analyses was focussed on interactions that were deemed sensitive and high risk, as these are likely to be the most relevant interactions to be considered at each site level assessment (Table A1.1). Interactions where there was insufficient evidence (IE) are not considered further here. These interactions will be considered in site level assessments where there is a known condition issue or further advice is received from JNCC or Natural England (Table A1.1). Where multiple sensitivities exist for features located across different bioregions, the most precautionary sensitivity has been displayed. Site-specific sensitivities will be used at the site level assessment stage.

Table A1.2 summarises the pressures of traps on designated features. It summarises all the interactions according to the key in Table A1.1.

The pressures listed in Table A1.2 are defined in JNCC AoO descriptions of pressures, based on Appendix 1 of the [UK Marine Pressures-Activities Database 'PAD': Methods Report | JNCC Resource Hub](#) (Robson et al., 2018).

Table A1.1. Gear/feature interaction sensitivity key. Pressures discussed within this review will be shown in red.

Key	
S	Indicates the feature is sensitive.
S*	Indicates the feature is sensitive to the pressure in general, but fishing activity/gear type is unlikely to exert that pressure to an extent where impacts are of concern (i.e. will be below pressure benchmarks).
IE	Indicates there is insufficient evidence to make sensitivity conclusions or a sensitivity assessment has not been made for this feature to this pressure.
NS	Indicates feature is not sensitive to pressure.
NS*	Indicates the feature is currently listed as not sensitive but JNCC and Natural England have advised that it should be considered further on a case-by-case basis at the site level.
NR	Indicates the pressure is not relevant for the gear type. There is no interaction between the pressure and biotope/species and/or no association between the activity and the pressure.

Table A1.2. Summary of the sensitivities of designated features to potential pressures from demersal seines and trawls, according to JNCC and Natural England Advice on Operations. Pressures discussed within this review are shown in red.

Potential Pressures	Designated Features											
	MCZ Species			Rocky Reef				Biogenic Reef	Annex I sandbanks and MCZ subtidal sediment habitats			
	Sea-pen and burrowing megafauna communities	Fan mussel	Ocean quahog	Fragile sponge and anthozoan communities	High energy circalittoral rock	Moderate energy circalittoral rock	Pink sea-fan	<i>S. spinulosa</i>	Subtidal coarse sediment	Subtidal mixed sediments	Subtidal mud	Subtidal sand
Above water noise	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
Abrasion or disturbance of the substrate on the surface of the seabed	S	S	S	S	S	S	S	S	S	S	S	S
Barrier to species movement	NR	NR	NR	NR	NS	S*	NR	NR	NR	NR	NR	NR
Changes in suspended solids (water clarity)	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
Collision ABOVE water with static or moving objects not naturally found in the marine environment	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
Collision BELOW water with static or moving objects not naturally found in the marine environment	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
Deoxygenation	S*	S*	NS	S*	S*	S*	S*	NS	S*	S*	S*	S*
Hydrocarbon + PAH contamination	IE	IE	IE	NA	IE	IE	IE	NS	IE	IE	IE	IE
Introduction of light	NS	NR	NR	NS	NS	IE	NR	NR	S*	IE	NS	S*
Introduction of microbial pathogens	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR

Potential Pressures	Designated Features											
	MCZ Species			Rocky Reef				Biogenic Reef	Annex I sandbanks and MCZ subtidal sediment habitats			
	Sea-pen and burrowing megafauna communities	Fan mussel	Ocean quahog	Fragile sponge and anthozoan communities	High energy circalittoral rock	Moderate energy circalittoral rock	Pink sea-fan	<i>S. spinulosa</i>	Subtidal coarse sediment	Subtidal mixed sediments	Subtidal mud	Subtidal sand
Introduction or spread of invasive non-indigenous species	IE	IE	IE	IE	S*	S*	S*	S*	S*	S*	S*	S*
Litter	IE	IE	IE	IE	IE	IE	IE	IE	IE	IE	IE	IE
Nutrient enrichment	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
Organic enrichment	S*	IE	NS	NS	S*	S*	IE	IE	S*	S*	S*	S*
Penetration and/or disturbance of the substrate below the surface of the seabed, including abrasion	S*	S*	S*	NR	S*	S*	NR	S*	S*	S*	S*	S*
Physical change (to another seabed type)	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
Physical change (to another sediment type)	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
Removal of non-target species	S	S	NR	S	S	S	S	S	S	S	S	S
Removal of target species	S	NR	NR	S	S	S	NR	NR	S	S	S	S
Smothering and siltation rate changes	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
Synthetic compound contamination	IE	IE	IE	IE	IE	IE	IE	NS	IE	IE	IE	IE
Transition elements & organo-metal contamination	IE	IE	IE	IE	IE	IE	IE	NS	IE	IE	IE	IE

Potential Pressures	Designated Features											
	MCZ Species			Rocky Reef				Biogenic Reef	Annex I sandbanks and MCZ subtidal sediment habitats			
	Sea-pen and burrowing megafauna communities	Fan mussel	Ocean quahog	Fragile sponge and anthozoan communities	High energy circalittoral rock	Moderate energy circalittoral rock	Pink sea-fan	<i>S. spinulosa</i>	Subtidal coarse sediment	Subtidal mixed sediments	Subtidal mud	Subtidal sand
Underwater noise changes	NR	NR	NR	NS	NS	IE	NR	NR	NR	NR	NR	NR
Visual disturbance	NR	NR	NR	NR	NR	IE	NR	NR	NR	NR	NR	NR